THE EFFECTS OF NUTRIENTS AND HYDROLOGY ON SHALLOW LAKE PLANKTON AT ATTENBOROUGH NATURE RESERVE, NOTTINGHAMSHIRE

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Abstract

Eutrophication results in the loss of conservation and amenity value from shallow lakes. Efforts have been made to restore shallow lakes by reducing the external nutrient loading. Removing nutrient-rich inflows can reduce nutrient loading but may alter lake hydrology. This thesis is primarily aimed at investigating the effects of a nutrient-rich river on water chemistry and plankton by comparing six shallow (<3.5 m) lakes that are connected to and isolated from the River Erewash, in the Attenborough Nature Reserve, Nottinghamshire, U.K.

Lakes that received River Erewash discharge had higher nutrient concentrations and higher phytoplankton biomass than those that were isolated from it. Turbid water was also found in a lake isolated from the River Erewash but with a nutrient-rich inflow stream. Lakes isolated from the inflows had abundant submerged macrophytes and clear water but other lakes were turbid and devoid of macrophytes. In lakes without inflows, cyanobacteria were proportionally more abundant. Lakes receiving nutrient-rich water were generally dominated by small chlorophytes and centric diatoms. Phosphorus concentrations had little effect on the phytoplankton, N and Si in the connected lakes, and zooplankton grazing and N in the isolated lakes, probably limited phytoplankton.

A mesocosm experiment found chlorophyll-*a* concentrations were reduced by the addition of silica, and that the biovolume of dinophytes increased in the mesocosms without nitrate addition. Total zooplankton biomass did not change significantly between treatments.

Summer floods reduced P concentrations in the connected lakes, suggesting that internal P loading was diluted and flushed out. Phytoplankton biomass was also lower during floods. Cryptophytes and diatoms dominated the phytoplankton and cyanobacteria were rare during flooding. A simulation of lake restoration by river diversion using mesocosms confirmed the importance of lake flushing for reducing internal loading. Diverting the River Erewash in order to reduce the nutrient loading to Attenborough Nature Reserve may be problematic in the short-term, because the lack of flushing may increase the effect of internal P loading and favour cyanobacterial growth. Reducing the external supply of N may further stimulate the dominance of cyanobacteria while P concentrations remain high.

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Abbreviations used in the text

Abbreviation	Definition
ANOVA	Analysis of variance
CA	Correspondence analysis
Ca^{2+}	Calcium
CCA	Canonical correspondence analysis
chl-*a*	Chlorophyll-*a*
Cl-	Chloride
Cond.	Specific conductivity
CS-strategist	Competitive-stress tolerant species
C-strategist	Competitive species
DCA	Detrended correspondence analysis
DO	Dissolved oxygen
K+	Potassium
meq	Milliequivalents
mg	Milligram
Mg^{2+}	Magnesium
N	Nitrogen
Na^{+}	Sodium
NH_4-N	Ammonium
NO_3-N	Nitrate
P	Phosphorus
PCA	Principle components analysis
RM-ANOVA	Repeated measures analysis of variance
R-strategist	Ruderal species
S.E.	Standard error
Si	Silica
SiO_3	Silicate
SRP	Soluble reactive phosphorus
S-strategist	Stress-tolerant species
STW	Sewage treatment works
t	Tonnes
TDP	Total dissolved phosphorus
Temp.	Temperature
TN	Total nitrogen
TP	Total phosphorus
TSS	Total suspended solids
WRT	Water retention time
z	Depth
z_{max}	Maximum depth
µm	Micrometer
L	Litre

Chapter 1 Literature Review

1.1 Introduction

The costs associated with the degradation of freshwater aquatic systems are substantial. Whilst ecological and aesthetic status and economic services to humans provided by the system are either lost entirely or compromised to some extent, increases in ecosystem productivity may provide some advantages. Directly or indirectly, eutrophication changes the economic value of aquatic ecosystems (Figure 1.1, Pretty et al., 2003). These changes are overwhelmingly detrimental, resulting in a net loss of value. Pretty et al. (2003) have estimated the annual cost of freshwater eutrophication in the U.K. to be £75-114.3 m. Loss of biological diversity is less easily quantified although it is an equally, if not a greater, driver of the numerous attempts that have been made to reverse or remediate the effects of eutrophication on lakes.

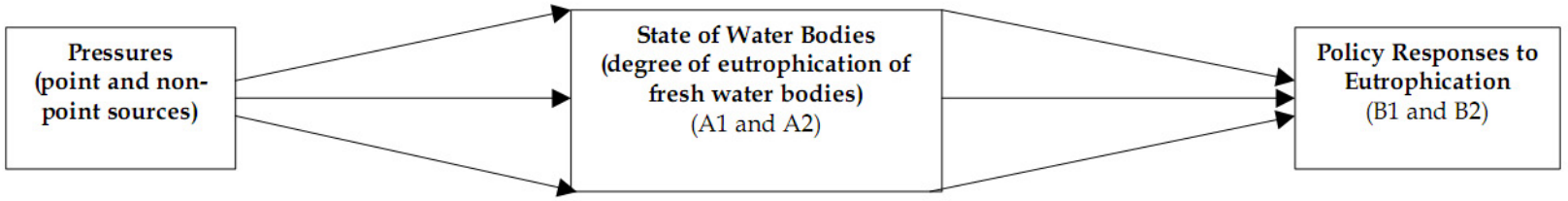
The process of eutrophication is generally well understood. The work of D. W. Schindler the late 1960s provided evidence that phosphorus (P) in particular was the regulating factor of freshwater phytoplankton production (Schindler et al., 1973; Schindler, 1978). Schindler's research has remained largely unchallenged and now forms the basis of our understanding of eutrophication. The Vollenweider model has contributed to eutrophication management by describing a linear relationship between chlorophyll-a concentration (an index of phytoplankton biomass) and P availability (Vollenweider, 1968) (discussed later from page 36).

In shallow lakes, the process of eutrophication begins with an increase in submerged macrophyte and periphytic algal biomass initiated by elevated P concentrations (Brönmark and Hansson, 1998). Macrophyte communities tend towards floating-leaved species in order to overcome increased water turbidity from increasing phytoplankton populations (Mason, 1996). Cyanobacteria (blue-green algae) and chlorophytes (green algae) become increasingly dominant components of the phytoplankton as its biomass increases (e.g. Schindler et al., 1973). Cyanobacteria present a toxic threat to lake users and animals that may come into contact with the

water (Moss et al., 1996a; Pitois et al., 2001). Cyprinid fish populations increase dramatically during eutrophication (Jeppesen, 1998). Their consumption of grazing invertebrates shifts the zooplankton community towards smaller, and therefore inefficient grazers, which reduces grazing pressure on the phytoplankton (Brönmark and Hansson, 1998). Increased sedimentation rates of senescent algae and their subsequent decomposition deplete water-column oxygen concentrations, potentially resulting in fish kills in particularly eutrophicated lakes (Brönmark and Hansson, 1998).

Although the processes that cause eutrophication are well understood, reversing the results has proven to be complex in shallow lakes. An extensive literature surrounds the restoration of degraded lakes and highlights the variety of responses that have been observed (Moss et al., 1996a). A literature review provides a discussion of the meaning of restoration and the importance of the alternative stable states model. A variety of examples of the responses of shallow lakes to nutrient reduction are given to illustrate important concepts. The roles of nitrogen (N) and silica (Si) as mediators of ecosystem response to P reduction is receiving growing attention. Based on a discussion of their importance, this review argues that their role in shallow lakes may be highly significant, and that understanding the influences they exert can improve the success of lake restoration projects.

This thesis aims to explore the role that N, P and Si may play in determining the biomass and composition of phytoplankton and zooplankton at Attenborough Nature Reserve, Nottinghamshire. The thesis also considers the effects of different hydrological regimes on the lakes caused by varying connectivity to the nutrient-rich River Erewash. As the River Erewash is currently being diverted from the Attenborough Nature Reserve in order to reduce the nutrient loading to the lakes connected to it, particular reference is given to the potential response of plankton communities to changes in hydrology and nutrient loading associated with the river diversion.



Point sources

i. sewage treatment plants (N and P into water);
ii. industrial plants (biological wastes, N and P);
iii. power plants (atmospheric N products, carried in rain to water bodies).

Non-point sources

i. agriculture (nutrients from inorganic fertilizers, animal wastes, soil erosion); gaseous emissions from soils
ii. aquaculture and fish farming;
iii. forest management (nutrient leaching);
iv. transport (atmospheric N products);
v. rural septic tanks;
vi. natural background sources.

A. Damage costs – the reduced value of clean or non-nutrient-enriched water

A1. Social damage costs

i. reduced value of waterside dweliings;
ii. reduced value of water bodies for commercial uses (abstraction, navigation, livestock watering, irrigation and industry);
iii. drinking water treatment costs (treatment and action to remove algal toxins and algal decomposition products);
iv. drinking water treatment costs (to remove nitrogen for human health and ecological reasons);
v. clean-up costs of waterways (dredging, weed-cutting);
vi. reduced value of non-polluted atmosphere (via greenhouse and acidifying gas emissions);
vii. reduced recreational and amenity value of water bodies for water sports (bathing, boating, windsurfing, canoeing), angling, and general amenity (picnics, walking, aesthetics);
viii. net economic losses for formal tourist industry;
ix. net economic losses for commercial aquaculture, fisheries, and shell-fisheries;
x. health costs to humans, livestock and pets.

A2. Ecological damage costs

i. negative ecological effects on biota (arising from changed nutrients, pH, oxygen), resulting in changed species composition (biodiversity) and loss of key or sensitive species.

B. Policy response costs – costs of addressing and responding to eutrophication

B1. Compliance control costs arising from adverse effects of nutrient enrichment

i. sewage treatment costs to remove phosphorus from point sources;
ii. costs of treatment of algal blooms and in-water preventative measures (biomanipulation, destratification, straw bale deployment etc.);
iii. costs of adopting new farm practices that leach fewer nutrients.

B2. Direct costs incurred by statutory agencies for monitoring, investigating and enforcing solutions to eutrophication

i. monitoring costs for water and air;
ii. cost of developing and implementing eutrophication control policies and strategies.

C. Benefits of nutrient-enriched (eutrophic) water

i. increased value of freshwater and marine fisheries;
ii. fertilisation effect on farmland;
iii. improved sources of food for wild birds.

Figure 1.1 Economic costs and benefits associated with freshwater eutrophication (Pretty et al., 2002).

1.2 Semantics

1.2.1 Restoration or rehabilitation?

There is an overwhelming trend in the scientific literature to describe any attempt at improving the ecological and chemical functioning of lakes as 'restoration'. For the majority of cases, however, this is somewhat misleading. Bradshaw (1996) draws attention to this confusing use of terminology. 'Restoration' implies a return to an original state, which in aquatic ecology is rarely (if ever) achieved (Figure 1.2).

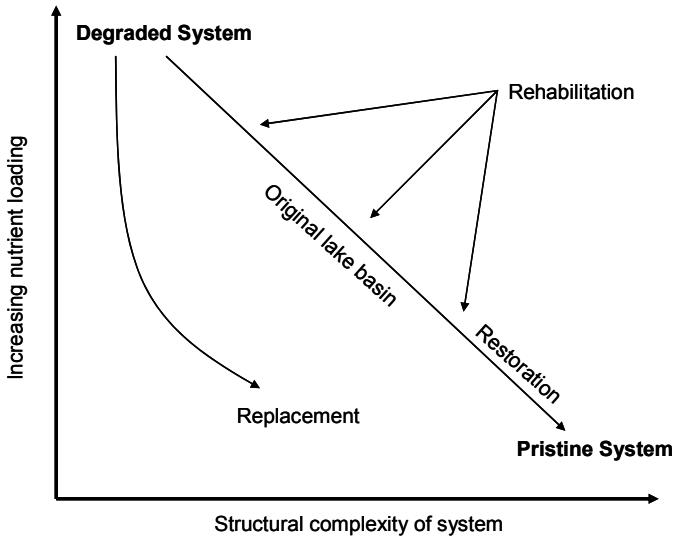


Figure 1.2 A graphical illustration of the differences between restoration, rehabilitation and replacement as functions of nutrient loading and structural complexity. Based on Bradshaw (1996).

'Rehabilitation' tends to be a far more realistic proposition for limnologists, as it does not imply a return to the perfection of a previous condition, but rather a return to a previous status. The picture is further confused when it is realised that the terms 'restoration' and 'rehabilitation' can be applied individually to different aspects of the system in question (for example a community, a habitat or species). The common perception that 'restoration' is an active human intervention can be challenged by natural processes which may restore ecosystems to, for example, bare soils after ice sheet retreat (Bradshaw, 1996) or saltmarsh after land reclamation.

Most attempts at shallow lake 'restoration' can be argued to be more representative of a process of 'rehabilitation'. Whilst complete restoration may be theoretically possible in some circumstances, and is arguably a more ethically desirable result (Bradshaw, 1996), various constraints (such as incomplete knowledge of the original state, economic factors and changes in the surrounding environment) mean it is rarely achievable for shallow lakes. Lake managers rarely attempt to create a 'carbon-copy' of the original state, but create conditions which are amenable to 'more representative' communities. Further complications arise from individual stakeholder interests in the purpose and use of the lake when planning restoration (Moss et al., 1996a). Despite its problems, the term 'restoration' will be used in its traditional limnological sense (i.e. the improvement of ecologically degraded ecosystems) throughout this thesis for convenience.

1.2.2 Resilience and resistance

The concepts of resilience and resistance are fundamental to our understanding of the problems faced when attempting to restore shallow lakes. Unlike the 'restoration'-rehabilitation' problem, there is little misuse of 'resilience' and 'resistance' although it is important to be aware of the subtle differences in meaning.

The property of resistance refers to the 'ability of a system to resist perturbation away from its steady state' (Carpenter et al., 1992) (Figure 1.3). Resilience is the rate of return to a steady state after a perturbation, and is considered as the sum of a variety of individual mechanisms that maintain ecosystem services (Carpenter and Cottingham, 1997). Quicker returns to the original steady state suggest greater resilience. A system may be resilient to change in both an ecologically desirable and valuable state, as well as in a degraded state (Carpenter and Cottingham, 1997).

The issue of resilience has been the subject of considerable debate in the ecological literature since it is pivotal in determining stability (e.g. Connell and Sousa, 1983; Sutherland, 1990; Peterson et al., 1998; Beisner et al., 2003). Likewise, defining whether a perturbation induces a significant or insignificant change in an ecosystem (Sutherland, 1990) is difficult. An analysis of literature discussing the distinction between 'resistance' and 'resilience' and the precise nature of perturbations would

merit a substantial discussion of its own. However, for the purposes of this review, it is sufficient to point out that the precise definition of resilience is open to various interpretations.

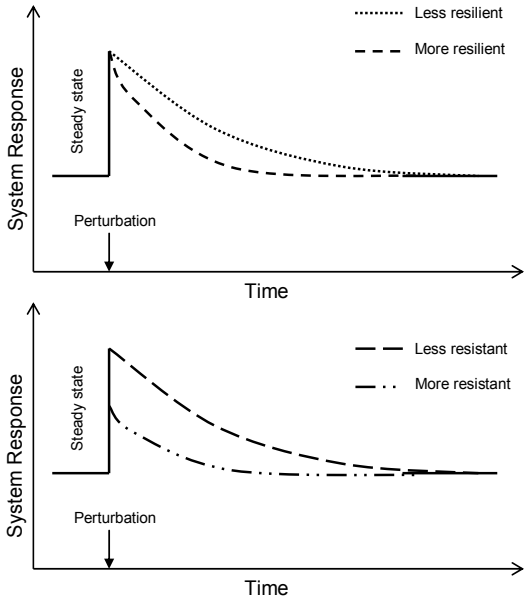


Figure 1.3 The differences between resilience (top) and resistance (bottom). Redrawn from Carpenter et al. (1992).

1.3 Alternative Stable States

Restoring high macrophyte biomass in shallow lakes is often seen as a measure of success in lake restoration. Submerged macrophytes are critical for the establishment of an ecologically diverse and valuable ecosystem (Jeppesen, 1998). The reasons for this are encapsulated in the alternative stable states theory. It is widely accepted that shallow lakes exist in one of two states - one dominated by phytoplankton, with depauperate submerged macrophyte communities and turbid water, and one with clear water, reduced phytoplankton communities and abundant submerged macrophytes (Scheffer et al., 1993; Moss et al., 1996a; Scheffer, 1998; Scheffer and Jeppesen, 1998; Scheffer, 2001; Carpenter, 2003).

Each state can occur at a wide range of nutrient concentrations. Pristine conditions are only likely to occur at total phosphorus (TP) concentrations of <25 µg P L^{-1}, whilst stable plant dominance in shallow water is likely to occur up to 50 µg P L^{-1} (Moss et

al., 1996a). At greater concentrations, submerged plant communities are likely to become less diverse and dominated by species such as *Ceratophyllum demersum* and *Potamogeton pectinatus*. Experimental evidence for reduced plant diversity is discussed below. There is an increased risk of a switch to the turbid state above 50 µg P L^{-1}, although the upper limit at which a system may not have the potential to exhibit either state is likely to be several milligrams of P per litre (Balls et al., 1989). Stability, as was alluded to earlier, is a result of the combination of resistance and resilience. This is illustrated by the 'stability landscape', or 'marble-in-a-cup' diagram in Figure 1.4. The system behaves in the same way as a marble does, by settling in 'valleys' of the 'landscape', representing a point of stable equilibrium. 'Hilltops' are unstable equilibria and also represent the division between one state and the next (Scheffer et al., 1993; Scheffer, 1998, 2001).

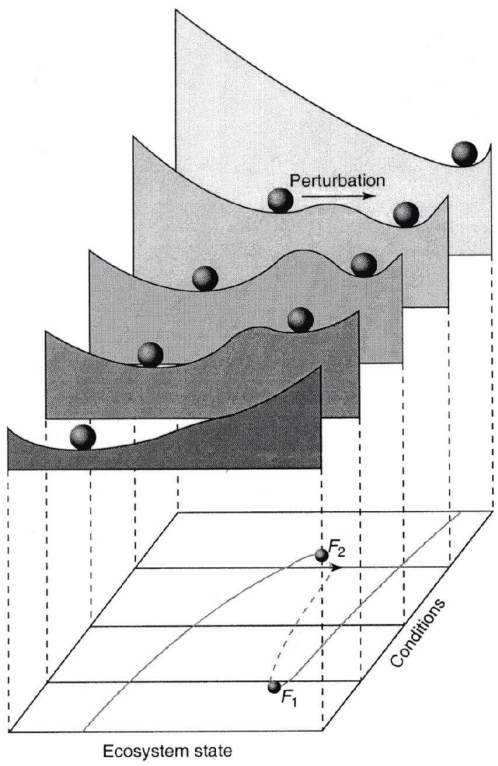


Figure 1.4 The alternative stable states model from Scheffer (1998)

The model also illustrates two potential mechanisms for a switch to occur. Movement of the 'marble' represents a disturbance to the ecosystem. Events that would cause this might include a fish kill or the destruction of submerged vegetation. Nutrient loading however affects the model's 'landscape', and therefore alters the 'effort'

required to shift the marble into the neighbouring 'valley'. Either equilibrium becomes increasingly unstable, since even a small perturbation could cause a switch when stability is low. Nutrient loading increases or decreases may be of sufficient quantity to shape the 'landscape' to the extent that only one 'valley' exists (Scheffer, 1998).

When a lake exists in a clear-water state, and has an abundant, stable submerged macrophyte community, a variety of positive-feedback processes operate which maintain water clarity (Figure 1.5). These processes alter the stability 'landscape' of the lake and prevent a change to the turbid, phytoplankton-dominated state. They can be categorised as either 'direct' or 'indirect' mechanisms (Jeppesen, 1998). All mechanisms create clear water and therefore increase the availability of light in order to further macrophyte growth and the stability of the system in a clear-water 'valley'. Direct mechanisms arise through the physical interaction of the plant with the surrounding water. Submerged macrophytes influence sedimentation and resuspension (Barko and James, 1998; Madsen et al., 2001) and reduce suspended solids concentrations (Van den Berg et al., 1998a; Havens, 2003; Horppila and Nurminen, 2003).

One of the indirect mechanisms by which submerged macrophytes create clear water is the 'refuge effect' which enables zooplankton to conceal themselves from predatory zooplanktivorous fish (Timms and Moss, 1984). Research in the Norfolk Broads, U.K. suggested that zooplankton congregate around floating-leaved macrophytes to avoid predation. Zooplankton migrated to open water during night time in order to graze on phytoplankton, a process known as diel horizontal migration (DHM) (Timms and Moss, 1984). The success of this refuge strategy for zooplankton is dependant on fish and macrophyte density (defined by PVI, percentage volume infested). Schriver et al. (1995) suggested that above a PVI threshold of 15-20%, *Daphnia* spp. became more resilient to high fish densities (CPUE [catch per unit effort] of 6). When CPUE was below 10 (approx. 2 fish m^{-2}), and macrophytes above 15-20% PVI, zooplankton grazing was sufficient to potentially consume more than 100% of the phytoplankton crop, suggesting that zooplankton were likely to be controlling phytoplankton growth. At low PVI (< 15-20%), and high fish density (>20 CPUE), potential grazing pressure was too low to be controlling phytoplankton growth (0.5-1% of phytoplankton crop) (Schriver et al., 1995). Perrow et al. (1999) argue that DHM as a survival strategy is

unsustainable. DHM is not sufficient to sustain *Daphnia* populations as fish feed in the dark. This means *Daphnia* populations will decline eventually even with abundant submerged macrophytes (Perrow et al., 1999).

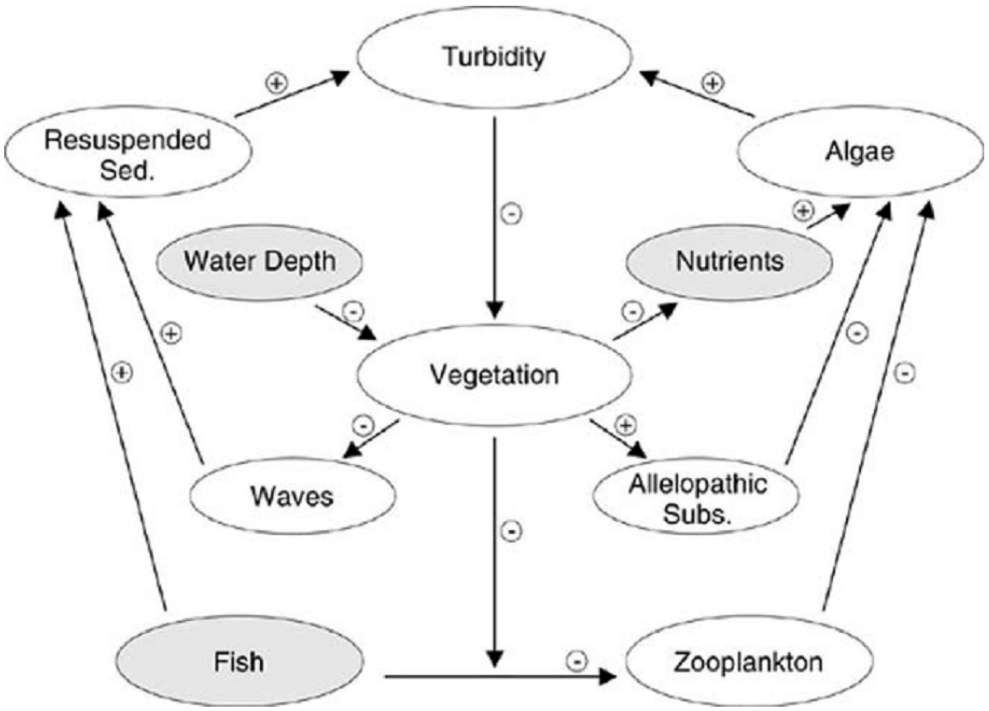


Figure 1.5 Feedbacks that may cause vegetation-dominated states and turbid states to be alternative equilibria. The qualitative effects of each route in the diagram can be computed by multiplying the signs along the way. This shows that both the vegetated and non-vegetated states are self-reinforcing. From Scheffer et al. (1993).

As macrophyte PVI increases during the year, macrophyte-associated grazers such as *Ceriodaphnia* spp. and *Simocephalus* spp. become the principle grazers. A PVI of 30-40% appears necessary to support these species (Perrow et al., 1999). However, this is dependent on fish community structure and biomass. The density of fish required to nullify the refuge effect reported by Perrow et al. (1999) was 1 m^{-2}, notably different from the 2 m^{-2} suggested by Schriver et al. (1995). This is most likely a consequence of roach (*Rutilus rutilus*) being dominant in the Norfolk Broads, which is a more efficient zooplanktivore than stickleback (*Gasterosteus aculeatus*) used in the experiments of Schriver et al. (1995).

Numerous studies have concluded that plants are capable of reducing phytoplankton abundance through the release of allelopathic substances. For example, laboratory experiments have suggested that *Myriophyllum spicatum* and *Ceratophyllum demersum* inhibited the growth of phytoplankton (Körner and Nicklisch, 2002). Different phytoplankton groups showed differing responses, with the *Oscillatoria* spp. being more affected by *M. spicatum* than chlorophytes and diatoms (Körner and Nicklisch, 2002). Laboratory microcosm, field surveys and *in situ* incubation experiments have suggested that the presence of *Stratoites alodies* (or exudates derived from it) reduces phytoplankton populations (Mulderij et al., 2006). Erhard and Gross (2006) suggest that *Elodea canadensis* and *E. nuttallii* reduced the growth of epiphytic algae, and cyanobacteria in particular. The chlorophyte *Chlorella vulgaris* was less sensitive to extracts of *Elodea* spp., whilst *Scenedesmus brevispina* increased in abundance, suggesting a degree of adaptation to the allelopathic effect of *Elodea* spp. (Erhard and Gross, 2006).

Epiphyte communities have been suggested to be a critical factor in determining the existence of a lake in either stable state. Phillips et al. (1978) used palaeolimnological evidence and microcosm experiments to investigate the effects of increased nutrient concentrations on epiphyte development. They presented a hypothesis that argued that the development of epiphytic communities on submerged macrophytes suppressed their growth, and that this occurred before phytoplankton blooms were evident. Epiphyte growth may be sufficiently luxurious to shade macrophytes and retard their development even if the water turbidity remains relatively low. Suppression of macrophytes then caused increased phytoplankton biomass and water turbidity, and the initiation of a positive feedback cycle which reduced macrophyte abundance (Phillips et al., 1978). It has been suggested that aquatic macrophytes may be capable of favouring an epiphytic community that is grazed preferentially by invertebrates, but evidence is lacking for any interaction. Based on replicated laboratory investigations, Jones et al. (2000) concluded that a relationship between macrophytes and their epiphytic communities was unlikely. Epiphyte communities did not encourage consumption by snails (e.g. by being nutritionally superior for invertebrates), and neither reproduction nor growth of grazing snails increased. Epiphyte communities differed between *Elodea* sp. and *Littorella* sp. treatments, although this difference was only slightly greater than that observed between inert artificial plants of differing

architecture. Grazing was by far the most significant factor affecting epiphytic communities (Jones et al., 2000). The density of grazing invertebrates, not nutrient concentration, was correlated with the density of epiphytes on aquatic macrophytes in plant-dominated lakes in Norfolk, U.K. (Jones and Sayer, 2003). Fish control of invertebrate populations was therefore suggested to be the major control of plant dominance and the source of stochasicity required by the alternative stable state model (Jones and Sayer, 2003).

1.4 Nutrient dynamics in shallow lakes

Attempts to reverse the effects of eutrophication were first applied to deep lakes, in both the USA and Europe (e.g. Lake Washington, Edmondson and Lehman, 1981; Lake Maggiore, de Bernardi et al., 1996) with encouraging results. The diversion of 99% of point source discharge from sewage treatment works (STW) away from Lake Washington in 1967 resulted in a decrease in chlorophyll-*a* and in-lake P concentration (Figure 1.6). Furthermore the proportion of cyanobacteria in the phytoplankton decreased dramatically (Edmondson and Lehman, 1981). Water quality in Lake Maggiore, Italy, declined during the 1960s and 1970s, accompanied by large phytoplankton biomass increases and cyanobacterial blooms (de Bernardi et al., 1996). P control measures, and a decline in industrial activity within the drainage area of the lake, resulted in a decrease in both phosphorus and chlorophyll-*a* concentrations from the early 1990s (de Bernardi et al., 1996).

Deep lake restoration successes, such as these, have relied on an oxic hypolimnion in order to immobilise P in sediments (see below). Deep lakes also tend to be located within landscapes where fewer nutrients are derived from diffuse sources than point sources. Point sources are those which are identifiable as a specific nutrient input to an aquatic system, and tend to be continuous and show little variability over time (Carpenter et al., 1998). However, non-point sources, although they may be continuous, are often episodic, and can be related to activities or events such as seasonal agriculture and heavy precipitation. Furthermore, diffuse sources may travel substantial distances. Consequently, they are considerably more difficult to manage

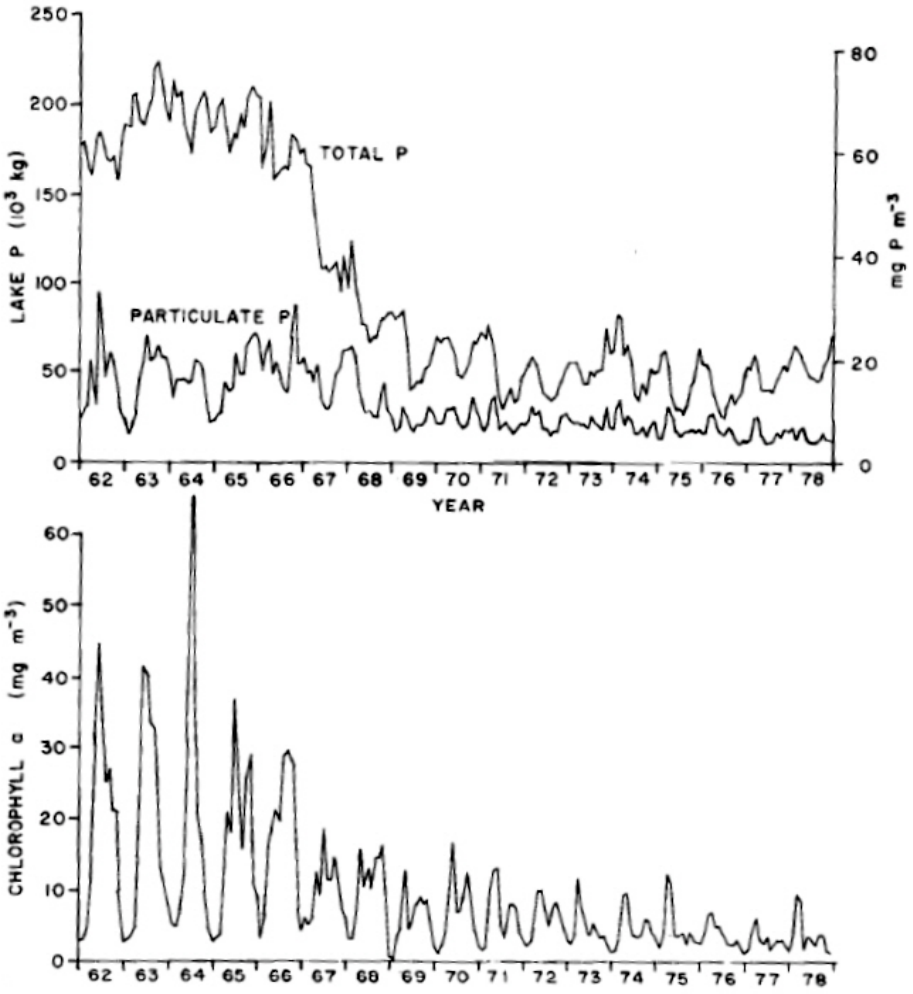


Figure 1.6 Total mass of P and particulate-P in Lake Washington and concentration of chlorophyll-a in surface water samples (from Edmondson and Lehmann, 1981).

and regulate (Carpenter et al., 1998). A large body of evidence, gathered particularly from shallow lakes in Denmark and the U.K., has developed to suggest that the response of shallow lakes to nutrient reduction is considerably more complex than deep lakes (e.g. Jeppesen et al., 1991; Søndergaard et al., 2000; 2001; 2003), not least because they are often subject to higher levels of diffuse pollution. This section provides a backdrop to a review of nutrient reduction projects through a discussion of the dynamics of three nutrients which are significant in controlling the functioning of shallow lakes: N, P and Si.

1.4.1 Phosphorus

In undisturbed temperate catchments, with abundant vegetation and forest, the supply of P to waterways is generally low (Kalff, 2002). Human influences, particularly from sewage discharge and agricultural sources such as fertilizers and livestock production (Figure 1.7), have released many lowland lakes from P limitation. As a result, it has received considerable attention as an agent of eutrophication. P concentration, in combination with other variables, forms the basis of the widely-used Orgnisation for Economic Co-operation and Development (OECD, 1982, in Dodds, 2002) classification of lake trophic status (Table 1.1). TP concentrations may also be a good predictor of chlorophyll-*a* concentration across a range of lake types, but considerable uncertainties can be associated with these estimates (e.g. Phillips et al., 2008).

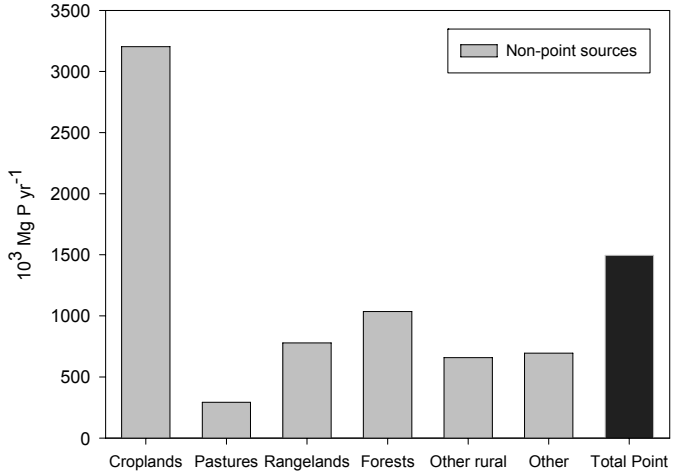


Figure 1.7 Sources of phosphorus discharges into the freshwaters of the USA. Based on Carpenter et al. (1998) (data from Gianessi et al. [1986] and Havens and Steinman [1995]).

Table 1.1: Boundary values for trophic classification according to OECD (1982, from Dodds, 2002).

Classification	TP (µg L^{-1})	Chlorophyll-*a* (µg L^{-1})		Mean Secchi depth (m)
		Mean	Max	
Ultraoligotrophic	<4	<1	<2.5	>12
Oligotrophic	4-10	1-2.5	2.5-8	12-6
Mesotrophic	10-35	2.5-8	8-25	6-3
Eutrophic	35-100	8-25	25-75	3-1.5
Hypertrophic	>100	>25	>75	<1.5

The removal of P from wastewater is straightforward. Tertiary treatment at sewage treatment works (STWs) utilises a variety of physical, chemical and biological techniques. Phosphate stripping involves the precipitation of P with lime and Fe (iron) or Al (aluminium) compounds, whilst biological techniques use organisms which store P within their cells and then the removal of these organisms from the system (Mason, 1996). P removal is inexpensive compared to N removal because P does not have an atmospheric component to its cycling (Kalff, 2002). A combination of these reasons make P management a favoured approach to water quality management and lake restoration.

P is deposited within lake sediments as either particulate or dissolved forms (detailed in Table 1.2). Particulate P is directly deposited, whilst dissolved P is incorporated into organic matter to form particulate P, before sedimentation (Søndergaard et al., 2001). P may become part of a variety of different compounds before either its permanent sedimentation and immobilisation or release to the water column. As part of the sediment, P incorporation into a variety of compounds is governed by the relative significance of different biological and chemical processes.

Many attempts have been made to relate inputs of P into a lake with the concentration of P within the water column in order to make some prediction of trophic state (Ahlgren et al., 1988). The idea that P exerts a control on trophic state originated principally from the work of R. A. Vollenweider, based on North American and European lakes (e.g. Vollenweider and Kerekes, 1980). Vollenweider's work was an important empirical addition to the scientific literature of its time, when the argument of P as a limiting factor (e.g. Schindler, 1978) was being advanced.

Table 1.2 Phosphorus forms and definitions applicable to freshwaters. Based on Moss et al. (1996).

Dissolved inorganic phosphate-P	Dissolved organic phosphate-P	P attached to colloidal clay or iron rich particles in suspension	P incorporated in living suspended algae and bacteria	P incorporated in dead detritus suspended in the water
Dissolved P		**Particulate P**		
Soluble reactive-P	P that may become available to plants and phytoplankton due to chemical and bacterial activity.			

Empirically derived steady-state models define the relationships between nutrient loading, sediment retention, lake water concentration and phytoplankton biomass and production (Ahlgren et al., 1988). These Vollenweider models typically take the form:

$$P_{lake} = \frac{L_p}{z(\rho_w + \sigma)}$$

where P(lake) is in-lake P concentration (μg L^{-1}); Lp, annual P loading (mg m^2 yr^{-1}); σ, a dimensionless unit of sedimentation; ρw, mean flushing rate (= *Q*/*V*, where *Q* = outlet discharge [m^3 yr^{-1}] and *V* = lake volume [m^3]) and *z* average depth (Ahlgren et al., 1988). Models of this type highlight the importance of depth in determining in-lake P concentration, and suggest that lakes with a higher flushing rate (lower water retention time, WRT) retain less P than those with a longer WRT. Vollenweider models may not be a reliable method of predicting P concentrations in lakes where P is retained in the sediments (Ahlgren et al., 1988).

For shallow lake restoration, empirical steady-state models may not accurately predict lake responses to nutrient loading reduction (Reynolds and Davies, 2001). Internal cycling and loading of P are significant processes in shallow lakes which provide a source of available P to phytoplankton (Jeppesen et al., 1991; Kleeberg and Kozerski, 1997; Nixdorf and Deneke, 1997; Søndergaard et al., 2001). The relationship between P loading and total phytoplankton biomass are not as strong as for deep lakes, and reductions in P loading may not necessarily correspond to a reduction of in-lake P concentrations as the Vollenweider model suggests. The magnitude of internal release of P after nutrient loading reduction from the sediments and the processes which govern it are a highly important when considering shallow lake restoration strategies (Scheffer, 1998).

P release from sediments is essentially the difference between sedimentation of particles and the upward P flux, as the sum of decomposition of organic matter and transport across the sediment-water column P concentration gradient (Søndergaard et al., 2001). It would therefore be logical to suggest that sedimentary P release would be related to the P concentration, although this correlation has not been shown to exist (Scheffer, 1998). Early research (e.g. Mortimer, 1941, 1942) highlighted the

importance of iron for its ability to prevent P release under aerobic conditions. The classic theory of P retention and release from lake sediments suggests that at the sediment surface, iron(II) is oxidised to iron oxyhydroxides, and P is precipitated from the water column or is adsorbed to iron(III) compounds (Scheffer, 1998; Søndergaard et al., 2003; Petzoldt and Uhlmann, 2006). When there is no oxidised microzone, and conditions are anoxic, P release can occur. In eutrophic, productive systems, anoxia is likely to occur as a result of bacterial degradation of organic matter (Figure 1.8, Scheffer, 1998).

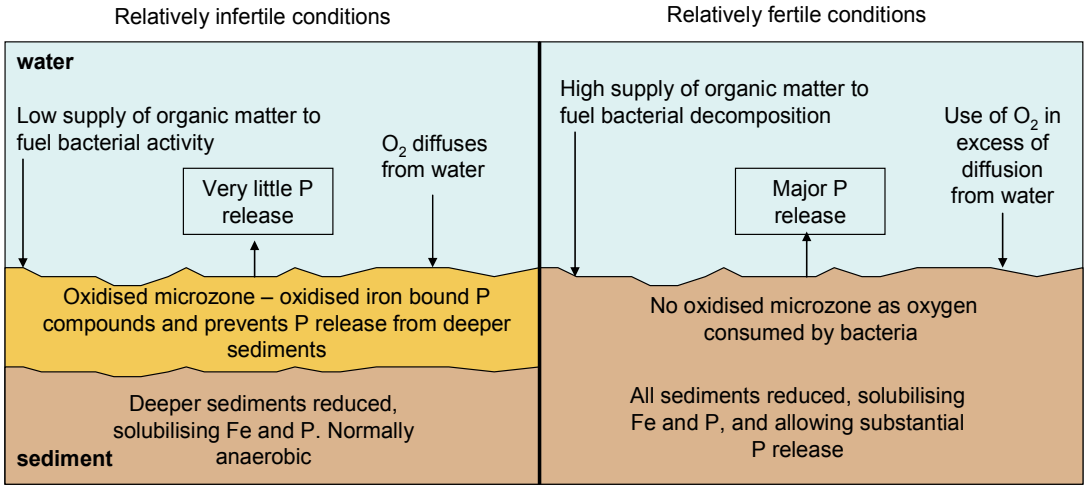


Figure 1.8 Some of the important phosphorus transformations that occur between sediment and freshwaters under relatively infertile (low productivity) and relatively fertile (high productivity) conditions. Based on Moss et al. (1996).

The role of iron as the principal determinant of P flux across the sediment-water interface has been challenged by research which suggests that microorganisms exert a significant control on P retention and release (Gächter et al., 1988; Gächter and Meyer, 1993). The traditional model of P flux (Mortimer, 1941, 1942) suggests that microbes play only a 'catalytic' role in the release of P by oxidising organic material, and since catalysts neither consume nor produce substances, microbes have been considered insignificant in comparison to chemical processes (Gächter and Meyer, 1993). However, microbes may influence P in a variety of ways, for example through the storage of P in sediments as refractory organic P in microbial biomass or the release of P during their decomposition (Gächter and Meyer, 1993). In oligotrophic

lakes, more P derived from organic detritus is stored by microbes as refractory P compounds than in eutrophic lakes, which may act as a stabilising mechanism to maintain low P concentrations (Gächter and Meyer, 1993). In a deep mesotrophic lake, microbial action was sufficient to cause organic detritus to accumulate soluble reactive-P (SRP, see Table 1.2 for explanation) during its settling and deposition on the sediment surface, suggesting that SRP could only be liberated to the water column if the rate of supply from underlying anaerobic sediments exceeded the rate of biological or chemical uptake and fixation on the sediment surface (Gächter and Mares, 1986).

It has been suggested that silica (Si) may interact with P cycling. Laboratory isotope labelling showed Si both displaced P bound in sediments and prevented further adsorption of P (Tuominen et al., 1998). Extrapolation of laboratory evidence into the field indicated that in eutrophic lakes with Fe-bound P, competition between Si and P may be responsible for significant mobilisation of P from the sediment surface to the water column. The mechanism is likely to be amplified in lakes with significant diatom blooms and high pH. Phytoplankton blooms increased pH, and at high pH, Si-induced P release is increased (Tallberg and Koski-Vähälä, 2001). The competitive ability of cyanobacteria is also higher at increased pH which may be partly responsible for fuelling the bloom with P release (Tallberg and Koski-Vähälä, 2001).

The addition of various salts to shallow lakes in order to precipitate P and seal it within sediments (‘phosphorus inactivation’) has received considerable attention as a method of overcoming internal P loading (Cooke et al., 1993). Al addition has had long term effects of reducing P concentrations (Cooke et al., 1993). Although Al dosing requires repeated small applications to maintain effectiveness, the precipitate formed is permanent (Lewandowski et al., 2003). In the hypereutrophic Lake Sønderby, Denmark, the addition of 31 mg Al m^{-2} in winter 2001 dramatically reduced in-lake P concentrations. Internal loading was reduced by 93% compared to pre-treatment years, and Secchi depth increased and submerged plants returned (Reitzel et al., 2005). Fe dosing has also been experimented with as a lake restoration tool, particularly since Fe is less sensitive to sedimentary redox conditions than Al (e.g. Deppe and Benndorf, 2002). It also mimics natural processes within shallow lakes (i.e. Fe-P binding) and is not toxic to plants as with Al (Moss et al., 1996a). The successes

of Fe additions are variable. Fe dosing had sufficient effect on internal P loading to induce a switch from phytoplankton to macrophyte dominance in a small drinking-water reservoir in southern England (Daldorph, 1999). However, the addition of iron aluminium sulphate to a deep, stratifying eutrophic lake and consequent reduction in SRP resulted in little change in annual populations of *Oscillatoria* spp. (Foy and Fitzsimons, 1987).

Nitrate nitrogen (NO_3-N) can also suppress the release of P from lake sediments. NO_3-N is an efficient oxidiser as it is more soluble in water than O_2 and can therefore penetrate further into the sediments (Hansen et al., 2003). This results in a larger pool of oxidised Fe being available for binding with P and the effective suppression of P release. Andersen (1982) found that in shallow Danish lakes, P release was prevented when NO_3-N concentrations were greater than 0.5 mg L^{-1}. The results of laboratory experiments and a whole-lake experiment by Foy (1986) showed that NO_3-N addition to lake sediments caused a delay in P release and reduced the amount of P diffusing into the water column. Jensen and Andersen (1992) found that the oxidised surface layer of sediments from Danish lakes increased in thickness when the concentration of NO_3-N in the overlying water was raised, reducing P release. Hansen et al. (2003) used a laboratory experiment to show that NO_3-N additions resulted in a reduction of P release from sediment cores and Stephen et al. (1998) found that P was scarce in mesocosms in Little Mere, U.K., where NO_3-N was added to the water.

Resuspension of lake sediments causes turbidity in shallow lakes by increasing suspended solid concentrations in the water-column. Particulate P can be resuspended simultaneously (Figure 1.9) and cause significant particulate P concentration change over short timescales. Soluble reactive phosphorus (SRP), and consequently TP, may or may not change under such circumstances as both P forms are also controlled by the response of phytoplankton to elevated P and the sediment-water equilibrium (Søndergaard et al., 1992 in Scheffer, 1998). Resuspension may cause a reduction in water-column P concentrations by adsorption to resuspended matter, although in eutrophic lakes this is unlikely (Scheffer, 1998). Ultimately, the balance between P desorption and adsorption depends on the relative saturation of sediments with P compared to water column P concentration (Scheffer, 1998).

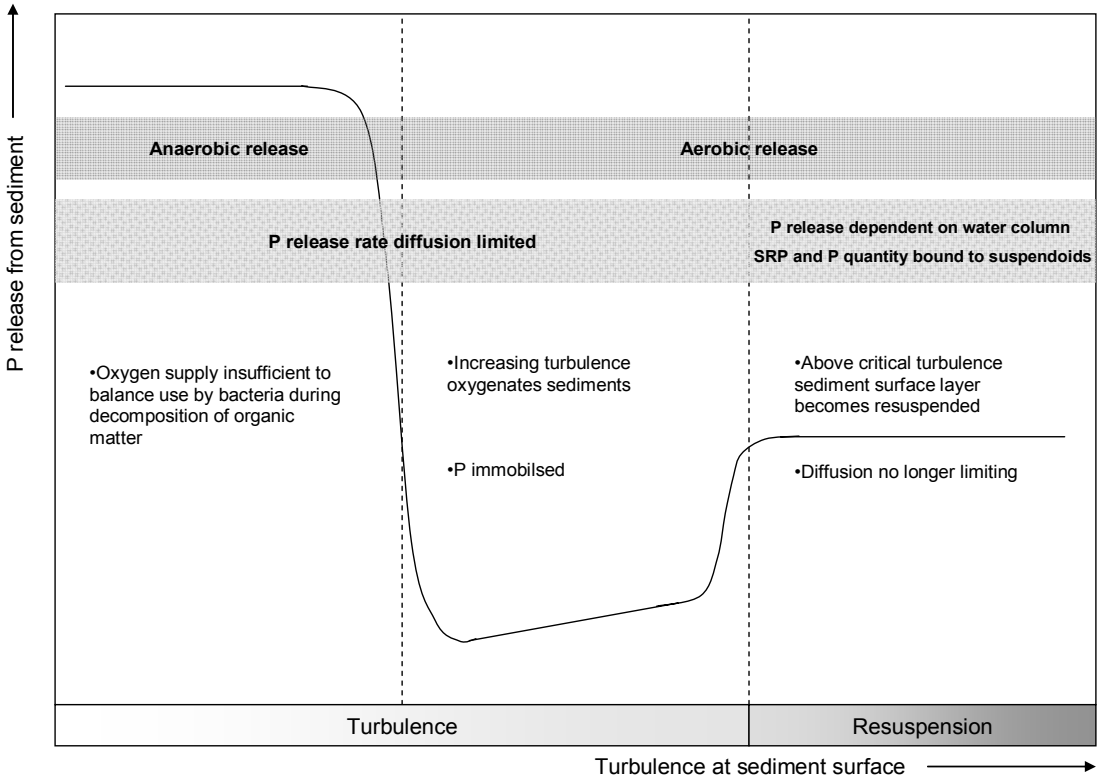


Figure 1.9 Schematic representation of the effect of turbulence at the sediment surface. Based on Scheffer (1998).

1.4.2 Nitrogen

Most N arrives into a lake basin from a range of diffuse sources including, although not limited to, arable agriculture, livestock farming and un-sewered urban settlements. The excessive application of fertilizers, either as artificial fertilizers or animal manure is a particularly important source of N to waterways (Carpenter et al., 1998). The diffuse nature of N sources (Figure 1.10) makes a reduction in loading substantially more complex than for point-source derived pollutants (Petzoldt and Uhlmann, 2006).

N is present in a variety of oxidised and reduced states in lake ecosystems (Kalff, 2002). These include nitrate (NO_3) and nitrite (NO_2), although the majority is present as organic N (Brönmark and Hansson, 1998). NO_3 may be reduced to NH_4 (ammonium) which is the preferred N source for phytoplankton and macrophytes

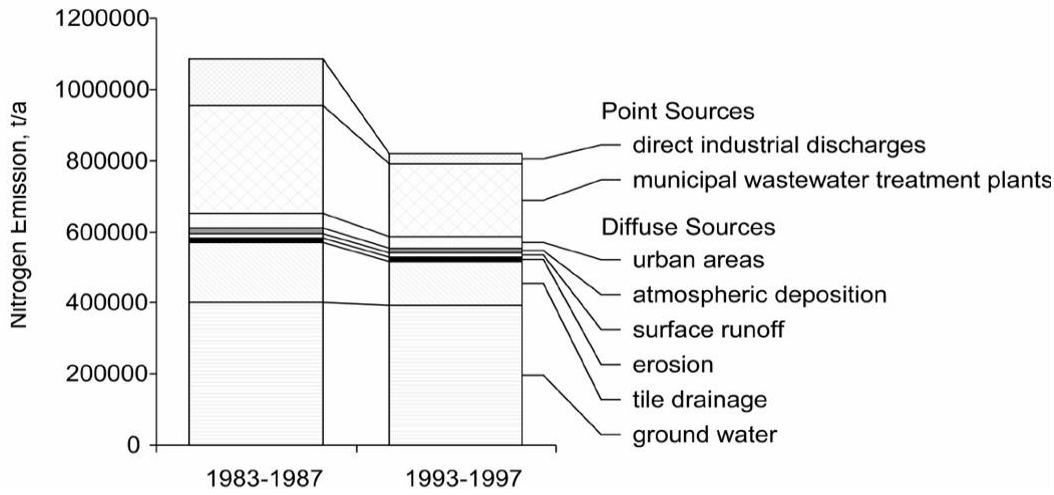


Figure 1.10 An example of contributors to nitrogen loading in a western European catchment (Germany). From Petzoldt and Uhlmann (2006).

(Brönmark and Hansson, 1998). The majority of NH_4 is derived from the breakdown of organic detritus (ammonification) (Kalff, 2002). Aquatic ecosystems appear to retain N as a function of their WRT, with lakes being intermediately retentive compared to wetlands and rivers (Saunders and Kalff, 2001b). The apparent retention of N occurs through denitrification, when NO_3 and NO_2 are reduced to produce gaseous N_2 and N_2O which is released to the atmosphere (Kalff, 2002). The uptake of N by macrophytes and the sedimentation of N-containing particulate matter are mechanisms of actual retention (Saunders and Kalff, 2001b).

Whilst the chemical transformations of N within freshwater systems are well understood, the ecological effects of N in shallow lakes and its implications in lake restoration have received less attention. Some pertinent points have been identified regarding the role it may play for phytoplankton communities and, more recently, aquatic macrophytes. For example, the cyanobacteria are a group of prokaryotic phytoplankton, some of which are able to utilise gaseous N_2 through specialised cells (heterocysts). Cyanobacteria assume a range of filamentous, colonial and single-celled forms (Wetzel, 1983; Kalff, 2002; Reynolds, 2006). Filamentous forms (such as species from the genera *Aphanizomenon* and *Oscillatoria*) have received particular attention from lake managers, as their distinctive appearance is aesthetically damaging and, some other forms produce toxins that are potentially fatal to humans and animals (Lawton and Codd, 1991; Pitois et al., 2001, Figure 1.11).

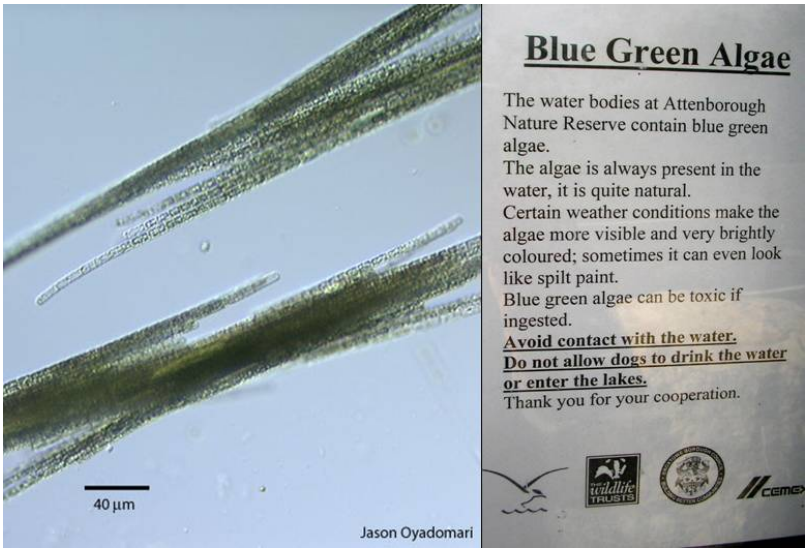


Figure 1.11 Left, *Aphanizomenon* viewed under a microscope (from http://www.keweenawalgae.mtu.edu/ALGAL_PAGES/cyanobacteria.htm). Right, warning sign at Attenborough Nature Reserve, August 2006.

Heterocystous cyanobacterial biomass has been shown to be low in systems where TN:TP ratio exceeded 29 to 1 (Smith, 1983) although this assertion has been challenged in shallow lakes (Jensen et al., 1994). A range of reasons exists for cyanobacterial success in aquatic environments, including their resistance to zooplankton grazing and competitive advantage at low light levels over eukaryotic phytoplankton (reviewed by Hyenstrand et al., 1998). The effect of TN:TP ratios on determining cyanobacterial success is significant since TP is traditionally controlled when implementing lake restoration efforts.

Several studies have suggested that N as well as P is important in determining ecological structure and functioning in shallow lakes. In the Cheshire Meres, U.K., N and P limitation were observed over different spatial and temporal scales (James et al., 2003). Bioassays suggested the majority of meres were N limited or co-limited by N and P during July, but not in October (James et al., 2003). N has been shown to be particularly important in the Qu'Appelle catchment in Canada in structuring total algal abundance of a chain of P-rich lakes connected by the Qu'Appelle River (Leavitt et al., 2006). N derived from urban sources was retained in the lakes, resulting in decreases in total abundance of algae with increasing distance downstream. Further studies on

the same lakes suggest that N-fixation by cyanobacteria increased with distance downstream. N-fixation in upstream lakes was negligible, although reached up to 77% of total N supply in downstream lakes (Patoine et al., 2006). In P-rich Lough Neagh, Northern Ireland, palaeoecological evidence suggests that historical algal abundance was more strongly correlated with inputs of diffuse N than P (Bunting et al., 2007).

In reservoirs in Kansas, USA, Dzialowski et al. (2005) found that the majority of reservoir phytoplankton communities responded only to additions of N together with P, implying co-limitation. The alleviation of light limitation strengthened the TN:TP and chlorophyll concentration relationship. In contrast to findings of James et al. (2003), in Kansas nutrient ratios accurately predicted limiting nutrients similarly to other published results (Table 1.3).

Table 1.3 Occurrence of nutrient limitation at differing molar TN :TP ratios reported by two studies.

Limiting nutrient	Guildford and Hecky, 2000	Dzialowski et al., 2005
N	< 20	< 18
P	> 50	> 65
N and P	20 - 50	20 - 46

In a survey of upland lakes in the U.K., co-limitation by N and P has also appeared to be the most frequent outcome based on bioassays (Maberly et al., 2002). However co-limitation was the most likely result when the dissolved inorganic N:total dissolved P (DIN:TDP) ratio was low. Cyanobacteria were not abundant when N was scarce which may be because low P concentrations and low pH may have inhibited their growth. In addition, the high flushing rate of the lakes surveyed is also likely to prevent cyanobacterial growth and therefore contribute towards N being more frequently limiting (Maberly et al., 2002).

Recent hypotheses and evidence draw parallels with terrestrial environments to suggest that elevated N concentration reduces aquatic plant diversity (Moss, 2001; González Sagrario et al., 2005; James et al., 2005). Using data from 60 shallow lakes located in Poland and the UK, James et al. (2005) showed that winter nitrate

concentration was the best predictor of plant species richness. *Ceratophyllum demersum*, *Potamogeton pectinatus* and *Lemna* sp. typified low-diversity communities. The most diverse plant communities were found at winter N concentrations of 1-2 mg NO_3-N L^{-1}, substantially lower than is likely to be attained under EU Water Framework Directive regulations (James et al., 2005). González Sagrario et al. (2005) used a series of mesocosm experiments to investigate trophic structure and water clarity responses to N and P enrichment in Lake Stigsholm, Denmark. Only when both nutrients were added to the mesocosms were significant effects recorded on macrophytes (decrease) and phytoplankton (increase) biomass. At mean TP concentrations of 0.07-0.13 mg P L^{-1} and TN of ≤1.2 mg L^{-1}, enclosures remained clear and submerged vegetation biomass high. A switch to a reduced macrophyte biomass and reduced water transparency occurred at TP ≥0.2 mg L^{-1} and TN of 1.2-2 mg L^{-1}.

The results concur with a survey of small shallow Danish lakes by González Sagrario et al. (2005, Figure 1.12) where the majority of lakes (63%) exceeded the potential TN concentration threshold for low macrophyte biomass. Plant loss at TN concentrations of 1-2 mg L^{-1}, with moderately high TP, is probably due to the alleviation of N limitation, allowing for periphytic and phytoplanktonic shading of macrophytes. Fish abundance may be instrumental in determining exactly what concentration of TN is required to induce a change from a clear water to turbid state due to their feeding upon grazing invertebrates (e.g. Jones and Sayer, 2003). These results suggest that consideration of N concentrations during lake recovery may be of importance and could account in part for the often slow, or delayed, recovery of submerged macrophytes after nutrient reduction.

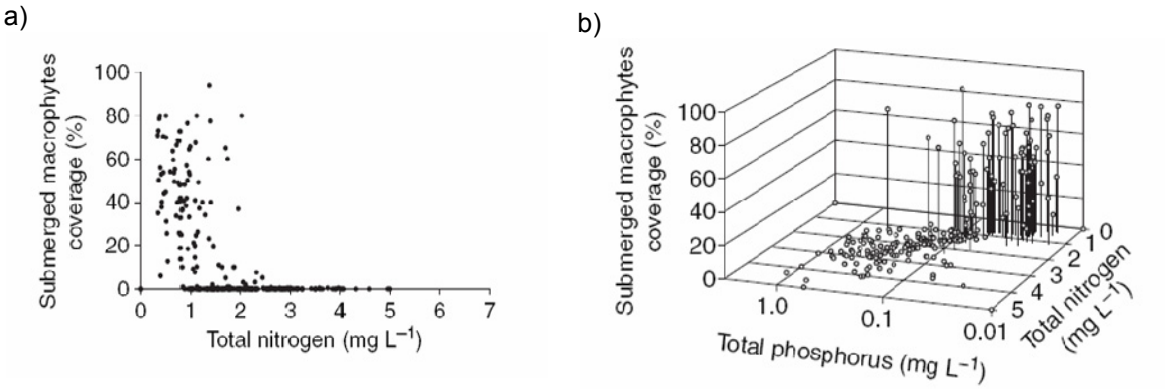


Figure 1.12 The relationship between submerged macrophyte coverage and (a) average summer TN and (b) average summer TP and TN based on 204 Danish lakes ($z_{average}$ = <5m, area >5ha). From Gonzalez-Sagrario et al. (2005).

1.4.3 Silica

Comparative to its abundance as an element in the earth's crust, silica (SiO_2) is scarce in aquatic systems since it is somewhat unreactive (Exley, 1998; Reynolds, 2006). SiO_2 in aqueous solution is derived from the hydrolysis and mechanical weathering of aluminium silicates (Reynolds, 2006). At a pH of below ~9, monosilicic acid (dissolved Si, DSi) is formed (van Dokkum et al., 2004; Reynolds, 2006). Anthropogenic releases of SiO_2 to the aquatic environment are insignificant at a global scale (amounting to <2% of Si from weathering processes) but can be significant locally (van Dokkum et al., 2004). These sources include detergents, water and wastewater production and paper pulp production (van Dokkum et al., 2004).

Although all species of phytoplankton require SiO_2 to some extent (Reynolds, 2006), it is the diatoms that exert the greatest influence on the cycling of SiO_2 in freshwater ecosystems. The only source of SiO_2 that can be utilized by phytoplankton and other biota is H_2SiO_4 (Exley, 1998; van Dokkum et al., 2004; Reynolds, 2006). Biogenic Si (also referred to as 'amorphous' or particulate Si), that forms the frustules of diatoms, is created from the condensation and polymerisation of DSi. This occurs on timescales of hours to days (Exley, 1998) and proceeds much more rapidly than the dissolution of diatom cells. Generally, the majority of Si in freshwaters repeatedly oscillates between

biogenic and dissolved forms unless it is removed from the system (e.g. through permanent burial, or loss to outflow).

During spring, phytoplankton in temperate lakes undergo exponential increases in population size over a period of a few weeks, initiated mainly by increasing temperature and light. Fast-growing *r*-strategist species, such as centric diatoms, often dominate during this period (Schmitt and Nixdorf, 1999). DSi is rapidly depleted by diatom uptake and is deposited as biogenic Si, a sequence well observed in both marine (e.g. Paasche and Østergren, 1980) and freshwater environments (e.g. Schelske and Stoermer, 1971; Schelske, 1999).

Bailey-Watts (1976) compared the quantity of biogenic Si in *Stephanodiscus rotula* populations with DSi concentrations in the water column of Loch Leven, Scotland. Diatoms are the primary constituent of the phytoplankton community in Loch Leven, a large, shallow ($z_{average}$ = 4m) and well mixed lake with a 'rich' supply of N and P. During the period of observation (1968-1971), populations of phytoplankton expanded and contracted rapidly, being driven largely (although not exclusively) by changing Si concentrations (Bailey-Watts, 1976). Interestingly, diatom maxima appeared not to show any relationship with Si concentration (Table 1.4). Confounding factors may include the uptake of Si by benthic algae, grazing by zooplankton and undetected losses of diatoms (such as through outflow or sinking) (Bailey-Watts, 1976).

Table 1.4 Dissolved Si concentrations at times of diatom maxima in Lough Leven (from Bailey-Watts, 1976).

Time of maximum		Contemporary DSi
Year	Month	(mg SiO_2 L^{-1})
1968	April	0.9
1969	March	6.0
	April	4.0
	May	0.5
1970	April	5.4
	September	0.5
	October	0.05
1971	March to April	0.25, 0.20
	September	3.3, 3.0

Schmitt and Nixdorf (1999) related the loss of planktonic diatoms in Lake Melangsee to a shift in the location of the majority of primary production. Si uptake appeared to shift from mainly pelagic to benthic habitats, caused by an increase in the quantity of light reaching the benthic environment. The observed decrease in Si concentrations in this eutrophic shallow lake could not be accounted for through assimilation by pelagic diatoms alone. Spring biomass of pelagic diatoms was usually less than 1.5 mm^3 L^{-1}, requiring only an estimated 0.45 mg SiO_2 L^{-1}, whilst Si actually reduced by 2-3 mg L^{-1} during this period. The frequent occurrence of resuspended benthic diatoms in sediment samples, increased oxygen in the surface sediments and high losses to outflow supported this primary production shift theory (Schmitt and Nixdorf, 1999). Reductions of P loading in Danish shallow lakes similarly induced a benthic diatom-mediated reduction in Si (Søndergaard et al., 2005).

The largest inputs to sediment Si reservoirs are often from biogenic sources and any abiotic inputs are generally less significant. Diatoms, due to their greater density compared to other phytoplankton, generally form a significant fraction of such biogenic material. Frustules may aggregate during sedimentation, increasing the rate of settling (Miretzky and Cirelli, 2004). Si sedimentation occurs mainly (although not exclusively) after diatom blooms and is therefore strongly periodic (Wetzel, 1983). For example, bimodal peaks of equal volume (June and November) in diatom sedimentation were observed in Lake Michigan following diatom blooms (Parker et al., 1977).

Sedimentation of diatoms can result in the permanent loss of Si from an aquatic system. In Lough Neagh, Northern Ireland, an estimated 26% of the inflowing Si was retained in the sediments due to diatom valve preservation as sub-fossils (Gibson et al., 2000). In deep lakes, Si can be effectively stored in deep waters and sediments and be unavailable for diatom uptake (Barbiero et al., 2002). The Laurentian Great Lakes have shown long-term decreases in epilimnetic Si concentrations due to phosphorus enrichment (Schelske and Stoermer, 1971; Schelske et al., 1983; Schelske, 1999). This process has been termed the ‘Si-depletion sequence’ (Schelske et al., 1983). The sequence begins with phosphorus-limited diatom production and little Si sedimentation. Alleviation of primary phosphorus limitation by anthropogenic enrichment induces rapid Si sedimentation, before a transition to secondary Si limitation (Figure 1.13).

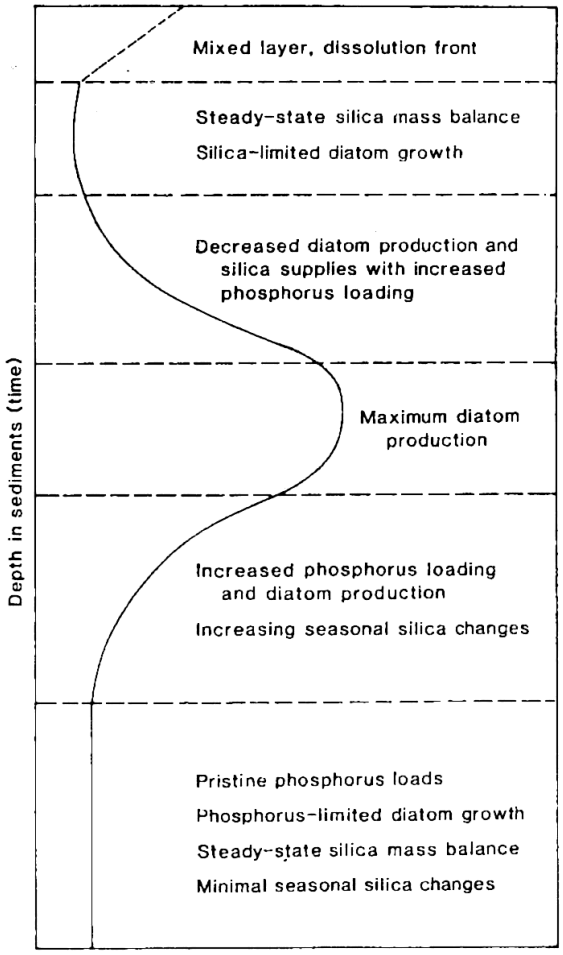


Figure 1.13 Hypothetical storage of biogenic silica in Great Lakes sediments based on phosphorus loading and the silica depletion sequence (from Schelske et al., 1983).

Recent evidence has demonstrated increases in Si concentrations following reduction of external P loading in Lake Michigan, USA. Babiero et al. (2002) report an increase in spring Si concentrations from 1.1 to 1.5 mg L^{-1} between 1983 and 2000, with a concomitant reduction in TP of approximately 1 µg L^{-1} over the same period. The large effects from such small P changes can be attributed to a likely greater reduction in bioavailable P than TP. In addition, the P requirements of diatoms are substantially less than for Si, reflected in the high Si:P ratios of diatom cells (Barbiero et al., 2002). A small change in the availability of P can therefore result in a proportionally greater change in Si assimilation, resulting in substantial change in Si dynamics. Schelske (1999) proposes three additional reasons why P enrichment has an greater influence on Si utilisation than nutrient ratios alone might explain:

1. P cycling is more efficient than Si cycling, as the dissolution of Si is a relatively slow process compared to the uptake, excretion and mineralisation of P,
2. 'accessory growth substances' (such as vitamins and trace metals) may be present in P-rich inputs, which further enhances the effects of nutrient enrichment,
3. nutrient (and accessory substance) addition stimulates the growth of diatoms at light levels which would usually be considered limiting.

As senescent diatoms descend through the water column, they are subjected to varying degrees of dissolution. Fauna may further influence dissolution processes, for example through zooplankton consumption (Ferrante and Parker, 1977) and chironomid activity (Gibson et al., 2000) (discussed below).

Dissolution of diatoms is a highly complex process and is critical for the replenishment of DSi in the absence of external sources. Dissolution may occur during the descent of senescent frustules through the water column or on the sediment surface. Zooplankton consumption and excretion of diatoms has been shown to speed up the settling rate of frustules in Lake Michigan (Ferrante and Parker, 1977). On the sediment surface, increasing pH, temperature, coarseness of sediment, grazing, bioturbation, water depth and exposure all increase the rate of dissolution (Flower, 1993). Salinity exerts a particularly strong control over diatom dissolution. Experiments conducted with natural sediment surface assemblages from Lakes Geneva and Nantua, France, by Barker et al. (1994), suggested that different salt solutions caused different rates of dissolution. At concentrations of 3.0 Mmol L^{-1}, the dissolution rate was higher in Li^{+}, K^{+} and Na^{+} solutions than for distilled water. Bivalent cation solutions (Mg^{2+} and Na^{2+}) of the same concentration (3.0 Mmol L^{-1}) caused lower rates of dissolution than for the monovalent solutions. Ryves et al. (2006) developed dissolution indices based on datasets from Greenland and the Northern Great Plains (NGP), USA. Their results suggested that dissolution rate was affected by salt type independent of salinity. Carbonate was important, but only for the NGP dataset. No lakes with carbonate concentrations above ~250 mg L^{-1} contained well preserved diatoms (Ryves et al., 2006).

Diatom frustules are highly adsorptive and remove metal ions such as Al from the water column (Exley, 1998) or sediment surface (van Dokkum et al., 2004) resulting in a reduction in the dissolution rate of the diatoms. The surface area and solubility of frustules decreases during their descent (van Dokkum et al., 2004). Laboratory studies have sought to quantify the effects of some of the variables that influence dissolution (e.g. Lewin, 1961; Rippey, 1983; Flower, 1993). Evidence from Lough Neagh sediment cores presented by Rippey (1977, 1983) demonstrated that temperature exerted the most significant influence on the rate of Si release. For example, an increase in temperature from 3 to 20˚C increased the rate of Si release by 260% (Rippey, 1977). In contrast, Spears et al. (2008) found that Si release was not affected by temperature, but increased light at the sediment surface reduced Si release, probably because of Si uptake by benthic algae. Published values of Si diffusion from sediment vary widely (Table 1.5).

Table 1.5 Published values of Si diffusion from sediments in freshwater systems (modified from Hofmann et al., 2002).

System	Lake area (km^2)	z_{max} (m)	Diffusion (g m^{-2} yr^{-1})	Comment	Technique*	Reference
Ursee (D)	-	11	6.1-12.3	Small bog-lake	LC	Tessenow, 1972
Grane Langsø (DK)	0.11	11.5	1.7	Oligotrophic	LC MB	Møller-Andersen, 1974
Kvindsø (DK)	0.14	2.5	7.3-69.3	Eutrophic flux is T dependent		
Loch Leven (Scotland)	13.3	mean = 4.0	19.4-144.6 (mean = 80.5)	Eutrophic	LC	Bailey-Watts, 1976
Lough Neagh (N. Ireland)	383	8.9	6.1-36.8	Eutrophic	LC	Rippey, 1983
Toolik Lake (Alaska)	1.5	25 (mean = 7)	0.2	Ultraoligotrophic, ice free 3 months a year	PWP	Cornwell and Banahan, 1992

*LC, laboratory cores: laboratory confinements of sediments in core tubes and measurement of solute concentrations in overlying water over time. PWP, pore water profiles: flux calculation from observed concentration gradients at the sediment-water interface based on Fick's first law and the molecular self-diffusion coefficient of the solute. MB, mass balance calculation.

The suggestion that benthic faunal activity increases the rate of nutrient release from sediments through bioturbation is long accepted, originating in the work of Tessenow (1964, in Gibson et al., 2000). Gibson et al. (2000) refined the concept of chironomid 'presence' into one of 'activity'. An arbitrary measurement of chironomid activity (biomass × respiration) produced a strongly seasonal pattern which agreed with the timing of sediment Si release in Lough Neagh (Gibson et al., 2000). No addition to the literature has expanded these concepts into a theoretical framework or modelled such processes in laboratory studies.

Despite a robust experimental understanding of factors influencing dissolution, applying these principles to the field is fraught with difficulty due to the 'multivariate' nature of diatom preservation (Flower, 1993). Ultimately, provided that pore waters of sediments are undersaturated with Si, a degree of dissolution will occur, whilst in high-energy, exposed settings diatom breakage is possible (Flower, 1993) and likely to speed up the dissolution of diatom cells. To appreciate the significance of Si liberation through dissolution it is important to set inputs and outputs to sediments in a temporal framework. Figure 1.14 shows the temporal changes over the annual cycle of Si

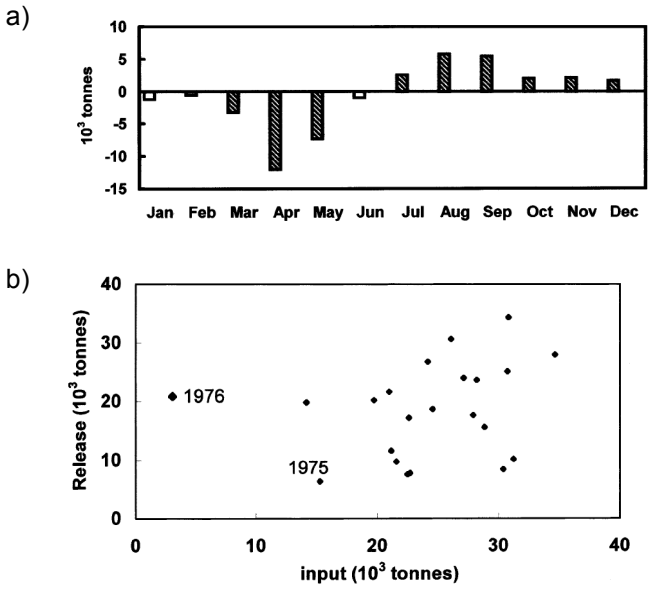


Figure 1.14 a) Calculated monthly average net sediment flux of SiO_2 1974-1997 (positive values are release) in Lough Neagh. b) Sediment input (March – June) and release (July – October) of SiO_2 1974-1997 (from Gibson et al., 2000).

sedimentation and release from Lough Neagh. Whilst a simple model of sedimentary output during late winter to spring appears to be compensated for by the storage of Si from early summer to mid winter, a more detailed consideration of sediment fluxes reveals a more complex pattern and lack of statistically significant input-output coupling on annual timescales. Of particular note are the values for 1975 and 1976, when low Si release in 1975 suppressed the spring 1976 diatom bloom, although the release of Si in 1976 was above the 1974-1999 average (Gibson et al., 2000). Long term averages, however, support a model where spring input corresponds strongly to sedimentary Si release.

Si limitation exerts an effect at both the species level initially and ultimately the physiological group level. Only species-level resource competition is 'mechanistically predictive' (Interlandi et al., 1999) although competition theory is often used to explain physiological group-level changes in phytoplankton communities (e.g. Krivtsov et al., 1999; Krivtsov et al., 2000a; van Dokkum et al., 2004, discussed below).

A classic study on the role of nutrients in shaping phytoplankton communities at the population level is that of Tilman and Kilham (1976; in Reynolds, 1998). They investigated the likely outcome of competition between the two diatom species *Asterionella formosa* and *Cyclotella meneghiniana*. Although consuming comparable quantities of P and Si, the affinity (ability to meet nutritional requirements at low external concentrations) of each species differed substantially. *Asterionella* showed lower affinity (indicated by half-saturation level; K_U) for P whilst *Cyclotella* had lower affinity for Si. This would enable *Cyclotella* to be a more successful competitor than *Asterionella* when Si is low, whilst the reverse is true when P is scarce. Field evidence from Lake Michigan echoed these findings (Tilman et al., 1982). A seasonal succession of diatom species has been hypothesised as a result of Si depletion by preceding species (Tilman et al., 1982, Figure 1.15).

Barberio et al. (2001) used ordination analysis to show a significant change in spring and summer diatom species composition as Si concentrations (and by implication Si:P ratios) increased due to ongoing reductions in phosphorus loading. Palaeoecological work from Lake Michigan (Stoermer et al., 1990) shows high abundance of high Si demanding diatoms in the surface sediments whilst species requiring less Si, such as

Cyclotella, achieved dominance when anthropogenic phosphorus loading remained high during the mid- to late 1800s.

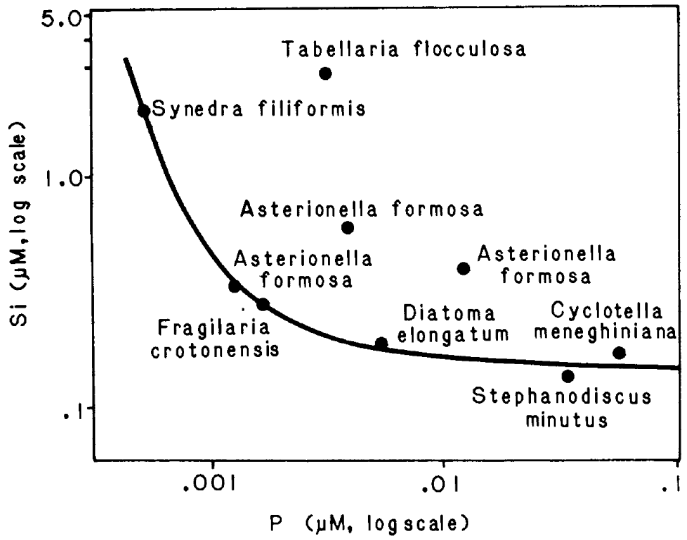


Figure 1.15 Interspecific tradeoffs in diatom competitive ability for silicate and phosphate at 20˚C. *Stephanodiscus* is the better competitor for silicate, whilst *Synedra* is a superior competitor for phosphate (from Tilman et al., 1982).

van Dokkum et al. (2004) chart a theoretical five-stage sequence of the effects of Si enrichment on physiological groups within phytoplankton communities. Firstly, spring and autumn blooms extend (1), with the possible effect of a reduction in chlorophytes and cyanobacterial blooms in summer due to the increased utilisation of phosphorus by the spring diatom bloom (2). This change is accompanied by (3) a shift in biomass from summer to spring and autumn, and (4) a possibility of a perennial increase in total phytoplankton. Finally, the effects of changes in algal community composition cascade through many trophic levels, since phytoplankton are at the base of many food webs.

Mathematical modelling produces results which reproduce this sequence. For example, the simple ‘Rostherne’ model was used to simulate the interactions between diatoms and cyanobacteria (Krivtsov et al., 1999; Krivtsov et al., 2000a). The model illustrated high Si concentrations increasing spring diatom populations with a consequent suppression of the summer cyanobacteria population caused by enhanced phosphorus uptake. Such processes, where one species influences the

next, was termed 'indirect succession regulation' (Krivtsov et al., 2000b) and was clearly an important mechanism in determining phytoplankton structure in aquatic ecosystems. Horn and Uhlman (1995) examined the switch in dominance between cyanobacteria and the diatom *Fragilaria* in the Saidenbach Reservoir, Germany. The originally mesotrophic reservoir (area 146 ha; z_{max} = 45 m) underwent anthropogenic eutrophication from the 1970s. Concomitant with increased nutrient loading was an increasing dominance of cyanobacteria over diatoms (Table 1.6). Despite an apparent transition towards non-siliceous species as Si became limiting, close correlations between Si:P ratios and the relative dominance of diatoms and cyanobacteria did not exist, although both Si and P controlled the phytoplankton community. N was considered unlikely to exert any influence since high agricultural loading maintains high N:P ratios (Horn and Uhlmann, 1995).

Table 1.6 Summary of nutrient and algal dynamics in the Saidenbach Reservoir, 1975-1995. SRP, soluble reactive phosphorus (from Horn and Uhlmann, 1995).

Date	Nutrient Conditions	Phytoplankton trends
1975 - 1980	Epilimnetic summer SRP < 4 µg l^{-1}.	P limited, nearly entirely diatoms. Few planktonic cyanobacteria.
1981 - 1986	Declining external Si loading. Diatoms limited by Si.	Steadily increasing cyanobacteria, subsequent to *Fragilaria* bloom. Occasional severe Si limitation favours earlier development of cyanobacteria population.
1987 - 1989	Higher Si and SRP inputs due to greater inflow.	Simultaneous abundant diatoms and cyanobacteria.
1990 - 1995	Drastic reduction in inflow volume and P concentration of inflow. Increased Si:P ratios, although Si at very low concentrations.	Diatoms become absent as Si limiting. Cyanobacteria become dominant phytoplankton component.

1.5 The importance of hydrology

Lakes are intimately connected to hydrological processes in their catchments (Winter, 2004). Precipitation, ground-water and rivers are the primary sources of water to lakes, although the significance of each source varies among lakes. Shallow lakes, which are more likely to be located in lowlands, may be connected to rivers and therefore river and stream discharge can be particularly important sources of water

(Winter, 2004; Reynolds, 2006). Large shallow lakes may lose considerable volumes of water to evaporation (Winter, 2004). Catchment-wide scale processes, such as precipitation, snow-melt and changes in surface permeability may alter surface runoff (Davie, 2002), and result in variations in the residence time of water within a lake (Winter, 2004). Hydrology is often overlooked in studies of shallow lakes. However, both nutrients and biology can be altered by variations in WRT.

Flooding events and reductions in WRT can increase or decrease nutrient concentrations in lakes. The response of lakes to variability in hydrology differs among studies. Alvarez Cobelas et al. (2006) found P concentrations decreased when precipitation increased groundwater discharge into a series of lakes. In a Polish floodplain, SRP concentrations were higher in lakes that were isolated from river inputs, although no clear trend in other nutrients was evident (Glińska-Lewczuk, 2009). In Lake Peipsi, Estonia, Haldna et al. (2008) found that P and NO_3-N concentrations were higher during a dry period. The effects of floods during spring reduced P and Si concentrations in floodplain lakes of the Daugava River in Latvia, but early summer flooding increased the concentrations of both nutrients (Paidere et al., 2007). Roozen et al. (2008) found increases in NO_3-N, P and Si associated with flooding in lakes along the River Waal, the Netherlands. In contrast, nutrient concentrations in a wetland in central Spain did not respond clearly to differences of water inputs (Angeler et al., 2000). The source of nutrients may explain some of these conflicting findings. Elliot et al. (2009) suggest that increases in streamflow increase the transport of diffuse nutrients to lakes, but not point-source nutrients. Increased river discharge may dilute the increased loading of diffuse source nutrients in lake basins (Elliott et al., 2009).

Theoretically, phytoplankton biomass is likely to decrease with increasing water inputs to a lake because of washout and dilution of cells. Washout is the loss of phytoplankton through outflows, and dilution is the reduction in the concentration of phytoplankton caused by an increase in the volume of water throughout which they are distributed (Reynolds, 2006). The realised change in phytoplankton abundance as a result of washout or dilution depends on the rate of flushing and the reproductive rate of the phytoplankton. When the difference between the loss of phytoplankton to outflow and phytoplankton reproduction are the same, then phytoplankton biomass is

less sensitive to changing flushing rate (Lucas et al., 2009). Lakes with smaller volumes and large catchments are more sensitive to inflow compared to large lakes with small catchments (Brook and Woodward, 1956), because the discharge to smaller lakes is proportionally greater compared to their volume (Reynolds, 2006).

Changes in phytoplankton biomass associated with hydrological regime have been the subject of several studies. Brook and Woodward (1956) concluded that the biomass of phytoplankton and the abundance of chlorophytes and cyanobacteria was lower in Scottish lakes with short WRTs than those with long WRTs. Dickman (1969) found that slower-growing species of phytoplankton became more abundant in an isolated section of a well-flushed lake in British Columbia, Canada. Rennella and Quirós (2006) found that high flushing rates increased phytoplankton biomass in lakes of the Pampa Plain in Argentina, because of increased nutrient availability. Longer WRT increased chlorophyll-*a* concentrations in Neuendorfer See, Germany, until the residence time reached 15 days, and decreased as residence time became longer (Walz and Welker, 1998). Phytoplankton decreased in response to short WRT in Mediterranean groundwater-fed lakes (Alvarez Cobelas et al., 2006). Water level increases in Lake Võrtsjärv reduced the total phytoplankton abundance, and caused a shift from the dominance of *Anabaena* sp. during low water levels to *Plantothrix* sp. at high water levels, reflecting changing light availability and an increase in the N:P ratio. In floodplain lakes of the Danube, increased water depths increased the importance of diatoms in the phytoplankton community at the expense of cyanobacteria (Mihaljević et al., 2009).

In addition to affecting the total biomass of the phytoplankton community, changes in hydrology may affect the composition of the phytoplankton. In highly disturbed environments, such as where flushing rates are high, some phytoplankton may have certain traits which favour their dominance over other species. For example, high rates of washout and dilution of phytoplankton in a lake may favour species with a rapid reproduction rate. The theoretical effect of different environmental conditions on phytoplankton assemblages has approached by extending Grime's (1973, 1977) *C*-, *S*- and *R*- strategies for terrestrial plants to phytoplankton. Each strategy represents an approach to surviving different combinations of stress and disturbance. Reynolds (1988) defines stress as 'nutrient limitation of the rates of cellular growth and

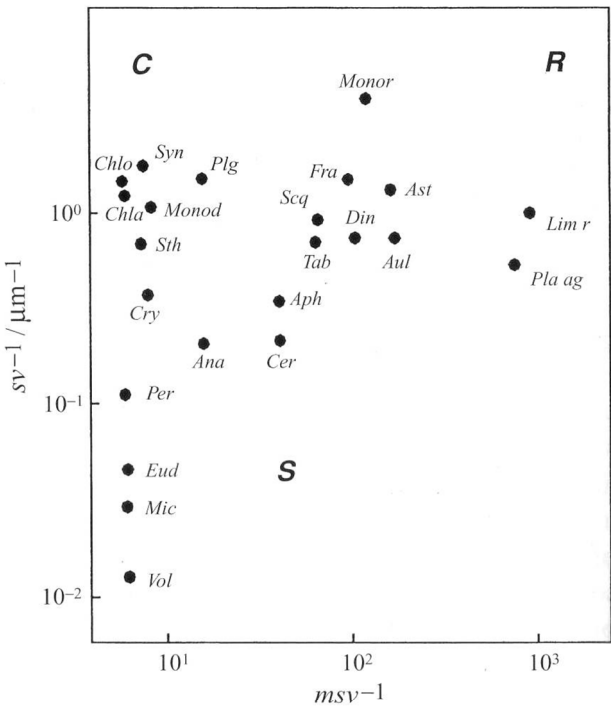
replication', and disturbance as 'frequent involuntary translocations of individuals out of the euphotic layer, especially if these result in the destruction of existing biomass'. Highly stressed and highly disturbed environments are not considered suitable for growth. *C*- strategists ('competitors') are able to rapidly exploit the available resources and have the highest growth rates in productive and undisturbed environments (Grime, 1977). *S*- strategists are slower growing species that can survive in stressful unproductive environments ('stress-tolerant'), and *R*-strategists ('ruderal') are adapted to highly disturbed and productive environments (Grime, 1977). Examples of typical species and characteristics are given in Table 1.7. However, most phytoplankton

Table 1.7 Summary of morphological and ecological traits of phytoplankton classified by Grimes's (1977) adaptive strategies, and representative species. Based on Reynolds (1988) with details from Reynolds (2006) and Jones and Elliott (2007).

Disturbance intensity	Stress intensity	
	Low	**High**
Low	**Competitors (*C*)** Mainly unicellular, low threshold for light and high surface to volume ratio. Highly susceptible to grazing by zooplankton and low rates of sinking. Includes *Chlamydomonas*, *Ankyra*, *Rhodomonas*.	**Stress-tolerant (*S*)** Some unicellular, many coenobial. Small surface to volume ratios, generally large. Adapted to low resource availability (e.g. nitrogen fixation). Most are motile, and can undertake controlled migrations. Includes *Microcystis, Anabaena*, *Ceratium*, *Peridinium*.
High	**Ruderals (*R*)** Some unicellular, many coenobial, high surface to volume ratio and high maximum linear dimension. High light requirements, some with high sinking rates, some susceptible to grazing. Includes *Aulacoseira*, *Asterionella*.	**None**

share traits of each strategy, for example *Scenedesmus* and *Monoraphidium* fall on a gradient between typically *C*- and *R*-strategists (Figure 1.16). These strategy traits have been found to correspond to natural phytoplankton assemblages under different regimes of disturbance. For example, García de Emiliani (1997) found that in a floodplain lake of the Parana River valley, periods of flooding were associated with increases in the abundance of *R*-strategists. Cryptophytes, which share traits of all three groups, also increased in abundance during flooding. When the lake became

isolated after the recession of the floodwaters, *C*- strategists, including *Scenedesmus* and *Monoraphidium*, were replaced with more slow-growing, *S*- strategist taxa, including dinophytes (*Peridinium*), and the cyanobacteria *Anabaena*.



Abbreviations:

Ana: *Anabaena flos-aquae*
Apha: *Aphanizomenon flos-aquae*
Ast: *Asterionella formosa*
Aul: *Aulacoseira subarctica*
Cer: *Ceratium hirundinella*
Chla: *Chlamydomonas*
Chlo: *Chorella* sp.
Chla: Chlamydomonas
Chlo: Chorella sp.
Cry: *Cryptomonas ovata*
Din: *Dinobryon divergens*
Eud: *Eudorina unicocca*
Frag: *Fragilaria crotonensis*
Lim r: *Limnothrix redekei*
Mic: *Microcystis aeruginosa*
Monod: *Monodus* sp.
Monor: *Monoraphidium contortum*
Per: *Peridinium cinctum*
Pla ag: *Planktothrix argardhii*
Plg: *Plagioselmis nannoplanctica*
Scq: *Scendesmus quadricauda*
Sth: *Stephanodiscus hantzschii*
Syn: *Synechococcus* sp.
Tab: *Tabellaria flocculosa* var. *asterionelloides*
Vol: *Volvox aureus*

Figure 1.16 Morphological properties of some freshwater phytoplankton in relation to their surface (µm^2) to volume (µm^3) ratio (sv^{-1}, µm^{-1}), and product of maximum linear dimension (m, µm) and sv^{-1} (msv^{-1}). Approximately spherical cells have an msv^{-1} value of 6 (from Reynolds, 2006).

1.6 Practical experiences of nutrient reduction

There is much evidence that suggests that the use of nutrient reduction as a technique for shallow lake restoration is complex and that the outcome may not be entirely as predicted. Whilst the Vollenweider model suggests that in-lake P concentrations will decline with external loading as a function of WRT, experiences in Denmark (Jeppesen et al., 1991) and globally (Jeppesen et al., 2005b) suggest that in lakes with short WRT (i.e. lakes that are well flushed), the historic accumulation of P outweighs the flushing of P from the lake. Seasonal differences in P release have

been shown to have consequences for in-lake P concentrations (e.g. Søndergaard et al., 2005). Furthermore, P release during late summer may fail to decline, even after considerable periods of time, such as in the case of the restoration of Barton Broad, Norfolk, U.K. (Phillips et al., 2005). Understanding seasonal variation in P release presents an opportunity for hydrological manipulation of lakes in order to aid nutrient reduction measures (e.g. Spears et al., 2006). If in-lake P concentrations are reduced and total phytoplankton biomass declines, changes in the structure of the phytoplankton have been observed, which can be of significant consequence if cyanobacteria increase. Nutrient reduction alone is often not enough to guarantee the return of submerged vegetation to shallow lakes and may require manipulations of the fish stock in order to bring about a stable and diverse macrophyte community (Moss et al., 1996a; 1996b). However, recent evidence suggests that rapid changes in foodweb structure can occur naturally after nutrient reduction and achieve a similar outcome to artificial manipulation (e.g. Jeppesen et al., 2005a). These issues are discussed in the following sections. An overview of the characteristics of the individual sites and datasets described are presented in Table 1.8.

1.6.1 In-lake P concentrations and WRT

Jeppesen et al. (1991) present a series of analyses of shallow eutrophic Danish lakes (TP ≈ 0.14 mg L^{-1}, chlorophyll-*a*, 0.051 mg L^{-1}) subject to high diffuse nutrient loading. In the majority of cases, the lakes underwent a 70% reduction in P loading. In most cases P concentrations had failed to reduce to levels expected after 4-16 years. This can be partly explained by the length of time required to dilute nutrient-rich water in the lake. Achieving a 95% reduction in in-lake P in a homogenous system with no sediment-water interactions, takes approximately three times the WRT (Sas, 1989 in Scheffer, 1998). Lakes with high WRT responded quicker than a simple dilution model would predict, attributable to the sedimentation and consequent storage of P in sediments. In contrast, lakes with a low WRT responded slower (10-300 times) than modelling predictions.

A dataset by Jeppesen et al. (2005b) covered lakes of varying depth, trophic state (7.5–3500 µg P L^{-1} prior to nutrient reduction), latitude and altitude. 5-10 years after nutrient loading reduction, in-lake TP concentrations where higher in the majority of

lakes than predicted by an equation relating TP to nutrient inflow and WRT, suggesting continuing internal loading. By 10-15 years after nutrient loading reduction, TP was closer to the equation's predictions. These results suggest that short-WRT

Table 1.8 Summary of case studies referred to in the text.

Lake Name / Location	Area Depth WRT	Nutrient reduction and other restorative measures	References
Various within Denmark (n = 27)	Mean = 20 ha Mean = 1.8 m Mean = 0.3 yrs	Various.	Jeppesen et al., 1991
Various within Denmark (n = 16)	Mean = 91 ha Mean = 2.5 m Mean = 0.19 yrs	Mean TP 0.322 mg L^{-1}, significant reductions in 4 lakes only.	Søndergaard et al., 1999
Global (n = 35)	3-355,500 ha, <5-177 m, 0.05 – 56 yrs	Pre-nutrient control = 7.5-3500 µg P L^{-1}. Various external measures used.	Jeppesen et al., 2005b
Augher Lough, N. Ireland	9.25 ha Max. = 14.5 m 0.46 yrs	Diversion of creamery effluent around lough.	Anderson et al., 1990
Various within Denmark (n = 8)	Mean = 539 Mean max. = 4.8 m Mean = 0.65 yrs	Start of P control = TP inlet, 0.556 mg P L^{-1} reduced to 0.126 mg P L^{-1} mg after 9 years. Various techniques.	Jeppesen et al., 2005a; Søndergaard et al., 2005
Mügelsee, Germany	730 ha Mean = 4.9 m 0.17 yrs	Catchment wide P measures to reduce external loading. Coincidental inflow reduction.	Köhler et al., 2005
Barton Broad, U.K.	60 ha Mean = 1.4 m 0.04 yrs	P removal at upstream STWs Sediment dredging after 17 yrs of P control.	Phillips et al., 2005
Alderfen Broad, U. K.	4.7 ha Mean = 0.8 m n/a	Diversion of inflow around Broad.	Moss et al., 1986, Moss et al., 1990; Perrow et al., 1994
Cockshoot Broad, U. K.	3.3 ha Mean = 1.0 m WRT not given	Isolation from river and sediment removal. Fish community removal 7 years later.	Moss et al., 1996b
Botshol Nature Reserve, The Netherlands	287 ha 1.5-3.0 m WRT not given	70% P reduction through chemical stripping at STWs and diversion of agricultural runoff.	Rip et al., 2005
Llandrindod Wells lake, Wales, U. K.	6.75 ha Mean = 1.25 m WRT not given	Biomanipulation of fish community and macrophyte reintroduction.	Moss et al., 2002
Loch Leven, U.K.	1332 ha Mean = 3.9 m WRT = 0.25-0.33 yrs.	Reduction in external loading (from 20-8 t TP yr^{-1}) 10 yrs prior to study.	Spears et al., 2006

lakes accumulated more P within their sediments and were therefore subjected to higher sedimentary P release, which outweighed the potential of their faster flushing rates to remove P (Jeppesen et al., 1991; 2005b). The experiences described here emphasise the caution of Ahlgren et al. (1988) that the application of Vollenweider

models to systems where P is retained is misleading and incorrect (see page 37). This is critical when considering shallow lake restoration and highlights the importance of a rigorous understanding of the timing and duration of P release from sediments after nutrient reduction.

1.6.2 Seasonal differences in P response

Eight years after nutrient loading reduction, Søndergaard et al. (1999) observed that annual retention of P was highest in a variety of lakes with lower P concentrations, although this decreased as summer TP increased. In common with other studies (Jeppesen et al., 1991; 2005b) retention of P was less than predicted by the Vollenweider model (see page 31 for details), particularly in lakes with high TP concentrations (> 0.2 mg L^{-1}). In lakes with summer TP below 0.1 mg L^{-1}, retention of P was negative only during July and August, although in lakes with summer TP >0.1 mg P L^{-1}, this period was extended to between April and September (Søndergaard et al., 1999). This can be explained as a result of an increase in the biological breakdown of organic matter, an enhanced rate of chemical processes that lead to a release of P and an increase in pH by a higher rate of summer-time photosynthesis. The spring clear-water phase led to reduced P retention in June, possibly as a consequence of reduced sedimentation of organic matter and an increase of benthic algal photosynthesis resulting in increased oxidation of the sediment surface (Søndergaard et al. 1995).

Søndergaard et al. (2005) examined seasonal responses of nutrients to reduced P loading. Late summer TP concentrations declined less than spring and early summer TP. This was likely to be a consequence of P available for release in spring and early summer being derived from the previous winter's sedimenting matter, whilst late summer P is derived historical P loading at greater depth in the sediments. During winter and particularly spring, P from deep sediments would be prevented from being released by oxidised sediment layers (Søndergaard et al., 2005).

During the restoration of Barton Broad, Phillips et al. (2005) reported a gradual shift in the magnitude and timing of P release from the sediments. Eight years after the implementation of nutrient reduction measures, P release occurred during May to July,

although by 19 years post-restoration the magnitude of this release had declined and occurred later in the year. Spring and early summer P concentrations were possibly related to catchment loading, with late summer and autumn P a consequence of internal loading (Phillips et al., 2005).

A consideration of the seasonal P dynamics in sediments and the water column suggests that manipulation of flushing could be used to reduce internal loading of eutrophic Loch Leven, Scotland (Spears et al., 2006). Late summer TP in the loch during 2004 was in excess of 80 μg L^{-1}. High water column TP coincided with low sediment TP and vice versa which suggested that much of the P contained in sediments was sensitive to release. June and July appeared to be the months of most retention whilst the largest gain to the sedimentary P pool occurred between December and February. Since the loch's hydrological regime can be controlled via sluice gates, there is the potential to coordinate maximum outflow with periods of high water column P concentration in order to assist nutrient reduction (Spears et al., 2006).

The temporal dynamics of P release are of great importance in understanding the outcome of nutrient reduction measures. This is particularly true if summertime P release continues, as this may be significant in maintaining high in-lake P concentrations and high phytoplankton biomass after restoration. Quantifying seasonal variation may enable the development of additional restoration measures based on carefully timed hydrological manipulation (e.g. Spears et al., 2006) to accompany nutrient reduction.

1.6.3 Phytoplankton community response

Despite nutrient reduction resulting in a decline in total phytoplankton biovolume, different algal groups have tended to respond differently to nutrient reduction measures. Jeppesen et al. (2005) observed that chlorophytes (as a proportion of total algal biomass) fell to <10% in shallow lakes with TP concentrations of <100 μg L^{-1} after nutrient reduction. As a proportion of total phytoplankton biovolume, diatoms, cryptophytes and chrysophytes increased. Diatoms may have been affected by a reduction in silica limitation (Jeppesen et al., 2005b). Diatoms also decreased in total

abundance in Barton Broad whilst shifting towards pennate forms away from small centric species. Increasing silica availability compared to P availability may have favoured larger pennate species (Sommer, 1988 in Phillips et al., 2005).

Palaeolimnological records have been used to investigate planktonic diatom responses to nutrient reduction and can be of considerable interest since they often provide information on longer timescales that can be derived than from monitoring data alone. Anderson et al. (1990) used a core taken from Augher Lough to show a clear response of planktonic diatoms to sewage management. When the lake became more eutrophic due to the onset of sewage disposal, the planktonic diatom community showed a switch from species indicative of mesotrophic conditions (such as *Aulacoseira* and *Asterionella*) to small *Stephanodiscus* species suggesting eutrophic conditions. When sewage effluent was diverted away from the lake, mesotrophic species returned. This is interpreted as a response to Si:P ratios, since *Aulacoseira* and *Asterionella* may be more dominant in high Si:P conditions (Figure 1.15), and could have been competitively advantaged by this when P was reduced by diverting the effluent. However, whilst chlorophyll-*a* and phosphorus data showed a rapid response to nutrient reduction, the planktonic diatom community continued to change, suggesting that their response was lagged and not in equilibrium with nutrient concentrations (Anderson et al., 1990).

In Barton Broad, spring and summer chlorophyll-*a* concentrations fell progressively at each five year interval after restoration as a result of declining TP. These changes were not observed in summer-autumn data until 15 years post-restoration. The response of cyanobacteria particularly is significant for lake managers. Filamentous cyanobacteria, which had been a significant component of the pre-restoration phytoplankton community in Barton Broad, remained proportionally abundant despite a decrease in their total abundance to near absence after 20 years (Phillips et al., 2005). Non-heterocystous cyanobacteria (such as *Limnothrix* and *Planktothrix* spp.) responded more quickly to nutrient reduction than heterocystous species (*Anabaena* spp. and *Aphanizomenon flos-aquae*), even though the summer N:P ratio increased. Zooplankton were not considered to be the cause of the overall decline in phytoplankton abundance in the broad, although they continued to create spring clear-water phases (Phillips et al., 2005). Jeppesen et al. (2005a) also observed that

despite a relative increase in the abundance of N, heterocystous cyanobacteria increased in biovolume during late summer and autumn, further corroborating the suggestion by Jensen (1994) that P may be more important in determining cyanobacterial species composition than N in shallow lakes. The same sequence of community change was observed in Mügelsee, with N-fixing species forming a higher proportion of phytoplankton biovolume, although in this case TN was reduced more than TP (Köhler et al., 2005).

1.6.4 Problematic macrophyte re-establishment

Although the return of submerged macrophytes is a highly important part of the restoration of shallow lakes, nutrient reduction measures alone often fail to allow recolonisation of vegetation. Studies often show a lack of reestablishment (e.g. Phillips et al., 2005), or a partial recovery and instabilities (e.g. Moss et al., 1990; Rip et al., 2005). Biomanipulation may be required to ameliorate conditions sufficiently for submerged vegetation to develop.

Oscillations between submerged macrophyte and phytoplankton dominance was evident after nutrient reduction measures in Alderfen Broad (Moss et al., 1990). A plant community of only *Ceratophyllum* spp. grew mainly in late July, which may not be sufficiently diverse to create the positive feedback mechanisms suggested by the alternative stable states hypothesis (see section 1.3 page 29, Moss et al., 1986). It is possible that alternations between phytoplankton and macrophyte dominance was caused by a build up of substances (such as sulphide) in the sediment that hindered plant growth. The four year period during which phytoplankton were dominant indicated the period required for the amelioration of conditions that hindered macrophyte growth (Moss et al., 1990). The increase in P release, as a consequence of increased detritus on the sediment surface, may have contributed to periods of phytoplankton dominance and have been exacerbated by the increased WRT (Perrow et al., 1994).

An inconclusive pattern of submerged macrophyte re-establishment was found by Jeppesen et al. (2005b) although in most instances colonisation was delayed. Reasons for this include a lack of a viable seedbank, hindrance by grazing waterfowl

and competition with benthic algae (Jeppesen et al., 2005b). The feeding habits of fish have been suggested to delay macrophyte growth. Roach (*Rutilus rutilus*) and perch (*Perca fluviatilis*) destroy leaves either by consumption or mechanical damage, whilst bream (*Abramis brama*) have been shown to stress macrophytes by sucking at the sediments (Körner and Dugdale, 2003).

Biomanipulation of fish stocks towards a reduction in fish that are zooplanktivorous or benthivorous species (including bream, roach and carp, *Cyprinus carpio*) has helped to improve macrophyte recovery in some shallow lakes (e.g. Moss et al., 1996b; Annadotter et al., 1999; Moss et al., 2002). The technique relies on the establishment of clearer water, principally as a result of an increase in grazing zooplankton biomass (Perrow et al., 1997a) although a reduction in plant damage and reduced suspended solids concentrations (Scheffer, 1998) are also of benefit to macrophyte growth. Despite several examples of success, some authors remain sceptical about its success as a technique for lake restoration (de Melo et al., 1992) whilst naturally occurring changes in fish communities post-restoration appear to achieve similar results (Jeppesen et al., 2005b) to labour-intensive and expensive biomanipulation.

1.6.5 Natural adjustments of food-web structure

Experiences of nutrient reduction suggest that food-web structure may change rapidly. In the case studies analysed by Jeppesen et al. (2005b), fish showed surprisingly fast responses to re-oligotrophication and this suggested that strong responses to nutrient reduction arise from a combination of top-down (predation) and bottom-up (resource) control. Fish community structure shifted away from cyprinid species and towards piscivores. This occurred with simultaneous reductions in total biomass (Jeppesen et al., 2005a). Increases in piscivores increased top-down control of phytoplankton in shallow lakes, evident by an increased zooplankton:phytoplankton biomass ratio, reduced chlorophyll-*a*:TP ratio (summer mean TP <100-150 µg L^{-1}) and a greater proportion of *Daphnia* spp. in the zooplankton (Jeppesen et al., 2005a). Incidental fish kills in Alderfen Broad, combined with poor recruitment, allowed the development of large-bodied zooplankton and contributed to creating clearer water (Perrow et al., 1994). Such changes appear not to be ubiquitous. The biomass of zooplankton (particularly of larger species) declined during the re-oligotrophication of Barton Broad,

suggesting that fish species remained largely zooplanktivorous. This may be a consequence of the wider connections of the Broads and river network of the area (Phillips et al., 2005). This result suggests that the effects of biomanipulation may occur naturally after in-lake nutrient concentrations are reduced. However, it is unlikely that relying on this as an aid to shallow lake restoration is a realistic option, given that no control can be exerted over the nature and extent of fish community change without direct intervention.

1.7 Literature Summary

Eutrophication has substantially damaged many freshwater aquatic systems, with a consequent loss in ecological and economic value and aesthetic damage (Moss et al., 1996a; Carpenter et al., 1998; Scheffer, 1998). The processes by which this occurs are well understood. Principles that have related P concentration to changes in lake ecology, in particular phytoplankton production (Vollenweider, 1968; Schindler et al., 1973; Schindler, 1978) have been particularly influential in shaping scientific approaches to the remediation of shallow lakes. As a consequence, the vast majority of shallow lake restoration attempts are founded on a reduction in the external loading of P. An increasing body of evidence shows that in the majority of lake restoration efforts, shallow lakes do not respond in line with modelled expectations (Ahlgren et al., 1988; Moss et al., 1990; Jeppesen et al., 1991; Søndergaard et al., 1999; Jeppesen et al., 2005a; 2005b; Søndergaard et al., 2005).

Understanding the dynamics of P in shallow lakes and the interactions of sediments with the overlying water column is critical to any shallow lake restoration project, since it is this which often accounts for the delayed reduction of phytoplankton production when external loadings are reduced (Scheffer, 1998). This phenomenon has been frequently observed in shallow lake restoration studies (Jeppesen et al., 1991; Søndergaard et al., 1999; Jeppesen et al., 2005b; Phillips et al., 2005). In order to determine both the quantity and time span of internal P loading, lake specific studies are required due to the many factors which may influence P release rates (including Fe binding, the role of microbes and sediment resuspension, Mortimer, 1941, 1942;

Gächter and Meyer, 1993; Scheffer, 1998) and their relative significance within individual basins.

N has received less attention in shallow lake restoration efforts. The role of N in eutrophication has generally been overlooked in favour of that of P. Since it is largely derived from diffuse as opposed to point sources (Petzoldt and Uhlmann, 2006), it is more difficult to manage external N loading than P loading. In addition, heterocystous cyanobacteria species are capable of N fixation and are able to overcome shortages of dissolved N. These species represent a significant challenge to lake management due to aesthetic damage to lakes and the health risks associated with their presence (Scheffer, 1998). Elevated N concentrations have been suggested by a number of authors to negatively affect submerged macrophyte communities (e.g. Moss, 2001; González Sagrario et al., 2005; James et al., 2005). Given that nutrient reduction measures are more frequently targeted towards P, this may in part account for some of the problems associated with macrophyte reestablishment during lake restoration in addition to, for example, fish foraging habits (Körner and Dugdale, 2003), grazing waterfowl, lack of seedbanks and benthic algal competition (Jeppesen et al., 2005b).

Si dynamics in lakes are principally controlled by the uptake of DSi by diatoms. In deep lakes this can lead to the permanent burial of Si (Schelske et al., 1983) whilst in shallow lakes oscillations of Si between biogenic and dissolved forms occurs due to the dissolution of diatom frustules. The release of Si from sediments has been demonstrated through laboratory experiments (Rippey, 1977, 1983). The close coupling of P enrichment, increased diatom production and Si retention in deep lakes has led to the development of the silica depletion hypothesis (Schelske and Stoermer, 1971; Schelske et al., 1983; Schelske, 1999). However, the applicability of this theory to shallow lakes is uncertain.

The practical significance of nutrient ratios in shaping phytoplankton communities remains an unresolved issue. Salient theoretical effects of nutrient ratios are given in Table 1.9. Field evidence of these patterns is often lacking, with no clear correlations between nutrient ratios and the dominance of phytoplankton groups evident (e.g. Horn and Uhlmann, 1995). During shallow lake restoration, nutrient ratios may alter significantly, for example, as a result of the diversion of nutrient-rich inflow followed by

sedimentary release of nutrients, or abatement measures which target one nutrient only. This may have significant consequences for phytoplankton communities and the success of a restoration project, particularly if prevailing nutrient conditions favour the development of cyanobacteria blooms. As Table 1.9, and the preceding discussion of the role of N (page 42) and Si (page 47) indicate, practical experiences of the effects of N:P and Si:P ratios are often not consistent with the theoretical expectations. Developing a deeper understanding of these experiences and expectations will be of significant consequence to future restoration attempts.

Table 1.9 Summary of the effects of nutrient ratios on phytoplankton communities.

Nutrient Ratio	Potential effects	
	High ratio	Low ratio
Si:P	Promotes growth of siliceous phytoplankton groups (e.g. diatoms). Diatoms with low affinity for P become more dominant within diatom community (e.g. *Synedra*, pennate species).	Promotes growth of non-siliceous phytoplankton groups (e.g. cyanobacteria, chlorophytes). Diatom species with low affinity for Si become more dominant within diatom community (e.g. *Cyclotella*, centric species).
N:P	No clear effect on abundance of cyanobacteria. P more likely to be limiting nutrient.	Promotes growth of both heterocystous (nitrogen-fixing, e.g. *Anabaena* and *Aphanizomenon*) and non-heterocystous cyanobacteria (e.g. *Limnothrix*). N more likely to be limiting nutrient.

1.8 Thesis aims

This thesis will use data gathered by monitoring a series of shallow lakes to quantify the effects of nutrient-rich river inflows on biological and chemical structure and functioning. The thesis will seek to relate chemical and hydrological regimes to biological communities and assess the importance of these for Attenborough Nature Reserve, by comparing data gathered from lakes of varying connectivity to the nutrient-rich River Erewash and contrasting nutrient enrichment. In addition to the dataset gathered by regular monitoring of the wetland complex, this thesis will use experimental evidence to further investigate how nutrient reduction efforts may affect the chemistry and ecological structure of shallow lakes. A mesocosm experiment will be used to manipulate the chemistry of lake water and the response of phytoplankton and zooplankton will be quantified. This part of the research will also enable an

analysis of internal nutrient dynamics, which is of key importance for future restoration efforts. Combined with the analysis of the monitoring dataset, this thesis will approach lake restoration from differing spatial and temporal scales and aims to provide insights into the likely response of the Attenborough Nature Reserve to the up-coming diversion of the River Erewash.

Therefore, the aims of this thesis are:

1. to compare and contrast the chemical and biological structure and functioning of the shallow lakes at Attenborough Nature Reserve isolated from and connected to the River Erewash,
2. to investigate how flooding events may affect the biology and chemistry of Attenborough Nature Reserve,
3. to experimentally simulate lake restoration by river diversion and explore the effects of nitrogen and silica on ecosystem community structure at Attenborough Nature Reserve,
4. to explore the likely effects of diverting nutrient-rich water from shallow lakes, and evaluate these effects in the context of the Attenborough Nature Reserve.

1.9 Thesis outline

This thesis has begun with an examination of contemporary and historical literature relating to shallow lakes. Key ideas relating to nutrient cycling in shallow lakes, and the historical provenance of development of these ideas have been traced to the current day. Attention has been given to the meanings of restoration in the context of shallow lakes. The causes of eutrophication, and the effects of the nutrients P, N and Si have been discussed and the role hydrology may play in determining phytoplankton community size and structure has been highlighted. With reference to lake restoration attempts, the difficulties encountered have been discussed in terms of the responses of chemical and biological components of lake ecosystems.

Chapter 2 (page 73) outlines the field site chosen for this study, and describes the development of the lake system to its present state. The results of previous research are synthesised to provide a backdrop to the thesis. Chapter 2 also provides an overview of the chemical and biological techniques used throughout the research.

Chapter 3 (page 91) details the chemistry and biology of the Attenborough Nature Reserve. The chapter considers the difference in the structure and functioning of the lakes and draws comparisons between those connected to, and isolated from, the River Erewash. Chapter 3 also provides a setting for subsequent chapters.

The next chapter (Chapter 4, page 197) explores the mechanisms that may control the total biomass of phytoplankton in Attenborough Nature Reserve. Chapter 4 uses correlation and multivariate analysis of the data presented in the Chapter 3 to explore potential controls of phytoplankton biomass and community structure.

Chapter 5 (page 224) is principally concerned with the hydrology of Attenborough Nature Reserve. The chapter uses data originally presented in Chapter 3 to compare the structure and functioning among rapidly and slowly flushed lakes, and to explore lake responses to extreme flood events. The chapter further investigates the role that hydrology might play in the restoration of shallow lakes.

Chapter 6 (page 285) reports the results of a mesocosm experiment. The experiment simulates lake restoration transferring water between connected and isolated lakes. Additionally, the experiment investigates how N and Si may interact to affect the plankton communities of shallow lakes.

In Chapter 7 (page 329), a synthesis of the key findings relating to the aims of this thesis is given. The chapter explores the implications of the findings of the previous chapters for the management of Attenborough Nature Reserve. Potential restoration efforts that may be complementary to using river diversion for lake restoration are presented and their potential use in Attenborough Nature Reserve are discussed.

Chapter 2 Study area and methods

2.1 Site Description

Attenborough Nature Reserve is located around 7 km south-west of the city of Nottingham in the East Midlands of the U.K (52° 53' N, 1° 14' W) adjacent to the River Trent (Figure 2.1). The six lakes which comprise the Attenborough Nature Reserve are among several in the area created by the extraction of gravel since the early 18th century (Godwin, 1923). The lakes of the Attenborough Nature Reserve were dug between 1930 and 1968 (Table 2.1). A summary of details regarding each lake is given in Table 2.2 and bathymetric maps of the lakes in Figure 2.2 (details regarding the data used for the bathymetric maps can be found in Chapter 5). Once the extraction ceased, the lakes were left to fill naturally. The land is owned by Cemex U.K. Ltd., and managed jointly with the Nottinghamshire Wildlife Trust and Broxtowe Borough Council. Much of the reserve is still used by the land owners for the transporting of gravel extracts from beyond the south-western boundary of the reserve to the processing plant (Figure 2.3).

The River Erewash flows 41 km from Kirkby-in-Ashfield in the north of Nottinghamshire in a predominantly southerly direction towards its confluence with the River Trent in the Attenborough Nature Reserve complex. Water from the River Erewash flows through three of the six lakes before exiting the lakes into the River Trent. The River Erewash forms a border with Derbyshire to the west along much of its length. The catchment covers an area of 206 km^2 (Natural England, 2009). Magnesian Limestone overlies the sandstones and mudstones in the Erewash valley. Coal is also a prominent component of the local geology, with deposits in the Erewash valley forming part of the Derbyshire and Yorkshire Coalfields (Natural England, 2009).

The Erewash catchment underwent intense urbanisation during the Industrial Revolution, particularly associated with the development of the railway network and the development of collieries, gas and tar works and blast furnace smelting (Environment Agency, 1995). As a result, diffuse and point-source pollution loading

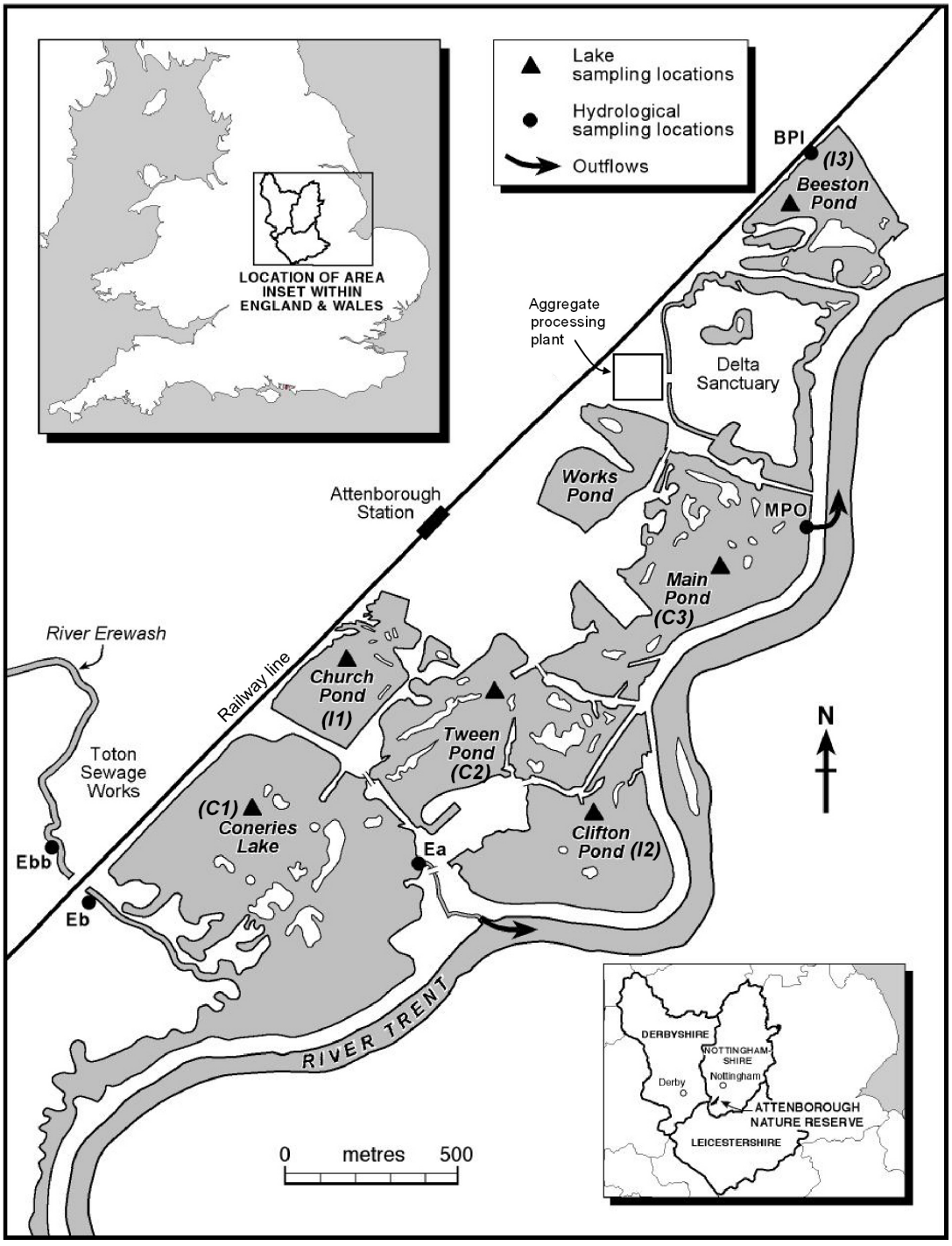


Figure 2.1 The Attenborough Nature Reserve, and insets, location in relation to local and national regions. Abbreviations and names are shown in Table 2.1.

Table 2.1 Local and abbreviated names of lakes, inflows and outflows

Local Name	Abbreviated name	Extraction period
Beeston Pond	I3	1941-1951
Beeston Pond inflow	BPI	-
Church Pond	I1	1962-1965
Clifton Pond	I2	1964-1968
Coneries Pond	C1	1966-1968
Erewash after lakes	Ea	-
Erewash before lakes	Eb	-
Erewash before Toton sewage treatment works	Ebb	-
Main Pond	C3	1939-1960
Main Pond outflow	MPO	
Tween Pond	C2	1956-1965

Explanation of abbreviations: **I** is used to denote lakes which are currently isolated from the River Erewash; **C** for lakes which are currently connected.

Table 2.2 Statistics of the lakes of Attenborough Nature Reserve. WRT is the mean for the period March 2005 to March 2008 and calculated for all connected lakes together . No WRT estimate is given for isolated lakes as the small inflows to these lakes were not quantified. Additional discussions regarding the hydrology of the lakes is given in Chapter 5 (page 224).

Lake	Area (ha)	Depth (m)		WRT (days)
		Mean	Max	
C1	51.7	1.4	3.3	13.8
C2	19.6	1.5	3.4	
C3	24.9	1.7	3.7	
I1	10.8	1.7	5.2	-
I2	14.7	1.1	3.6	-
I3	8.5	1.2	2.7	-

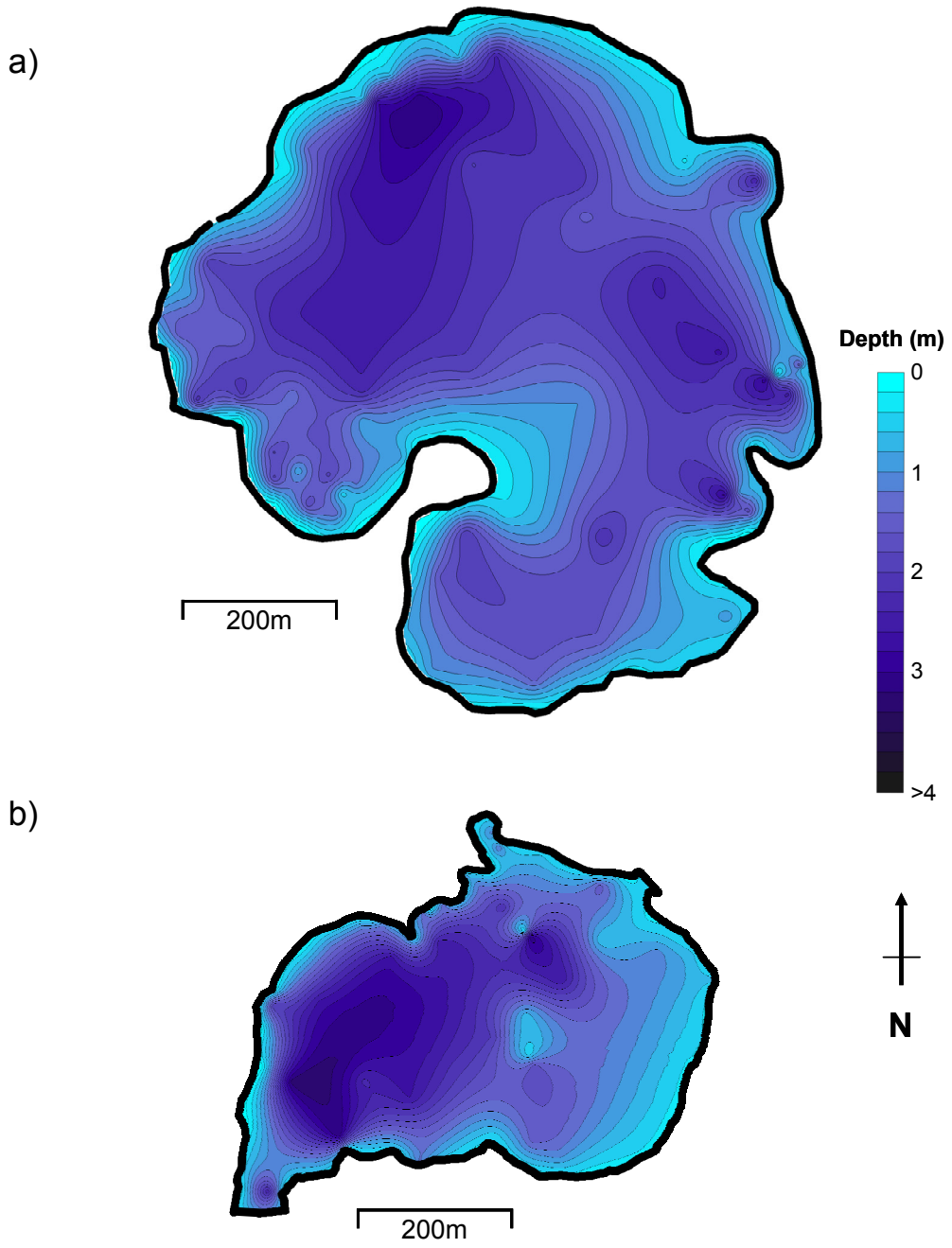


Figure 2.2 Bathymetric maps of the lakes of Attenborough Nature Reserve: a) Lake C1; b) Lake C2 (continued overleaf), c) Lake C3, d) Lake I1; e) Lake I2 and f) Lake I3. Note the variation in scale.

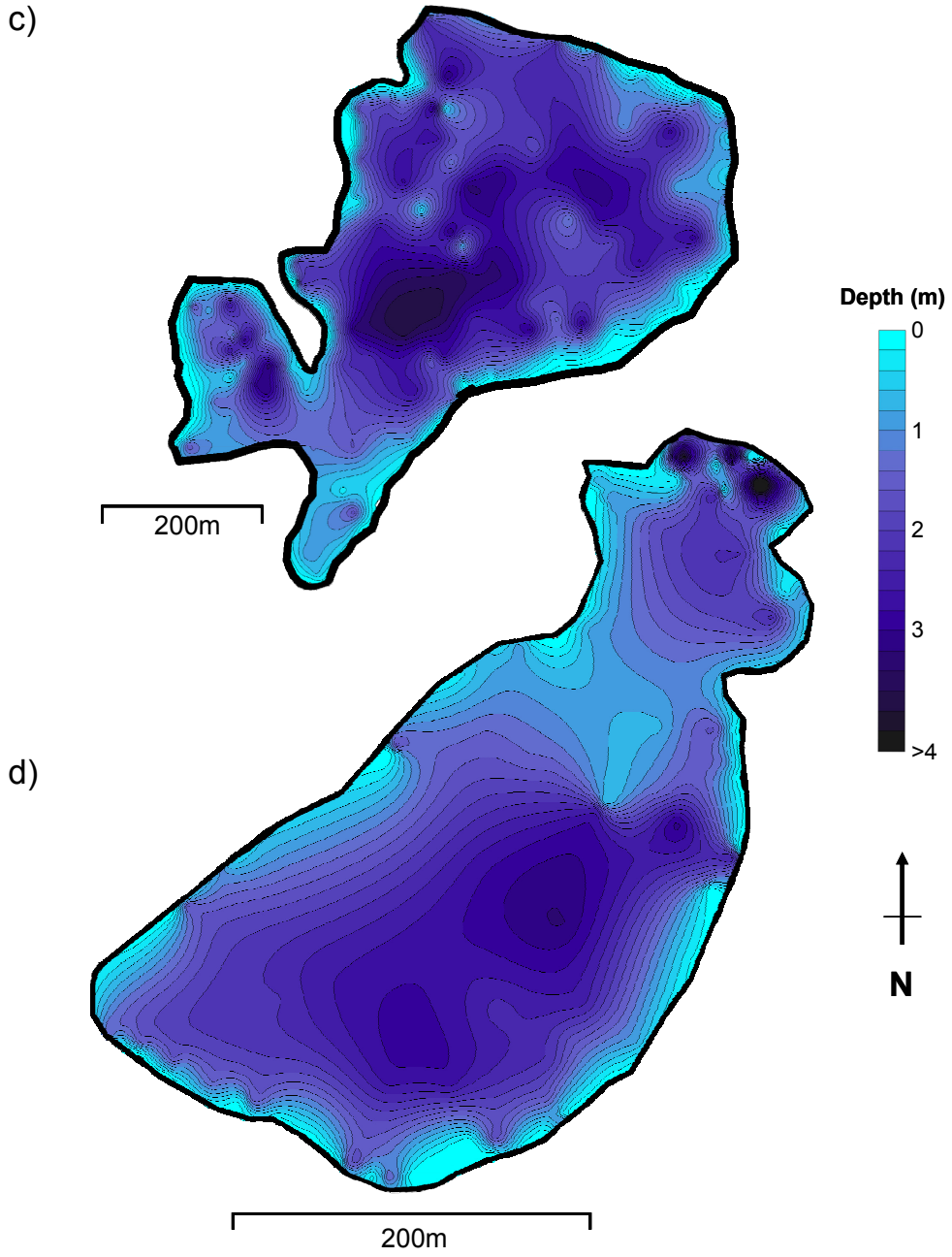


Figure 2.2 contd.

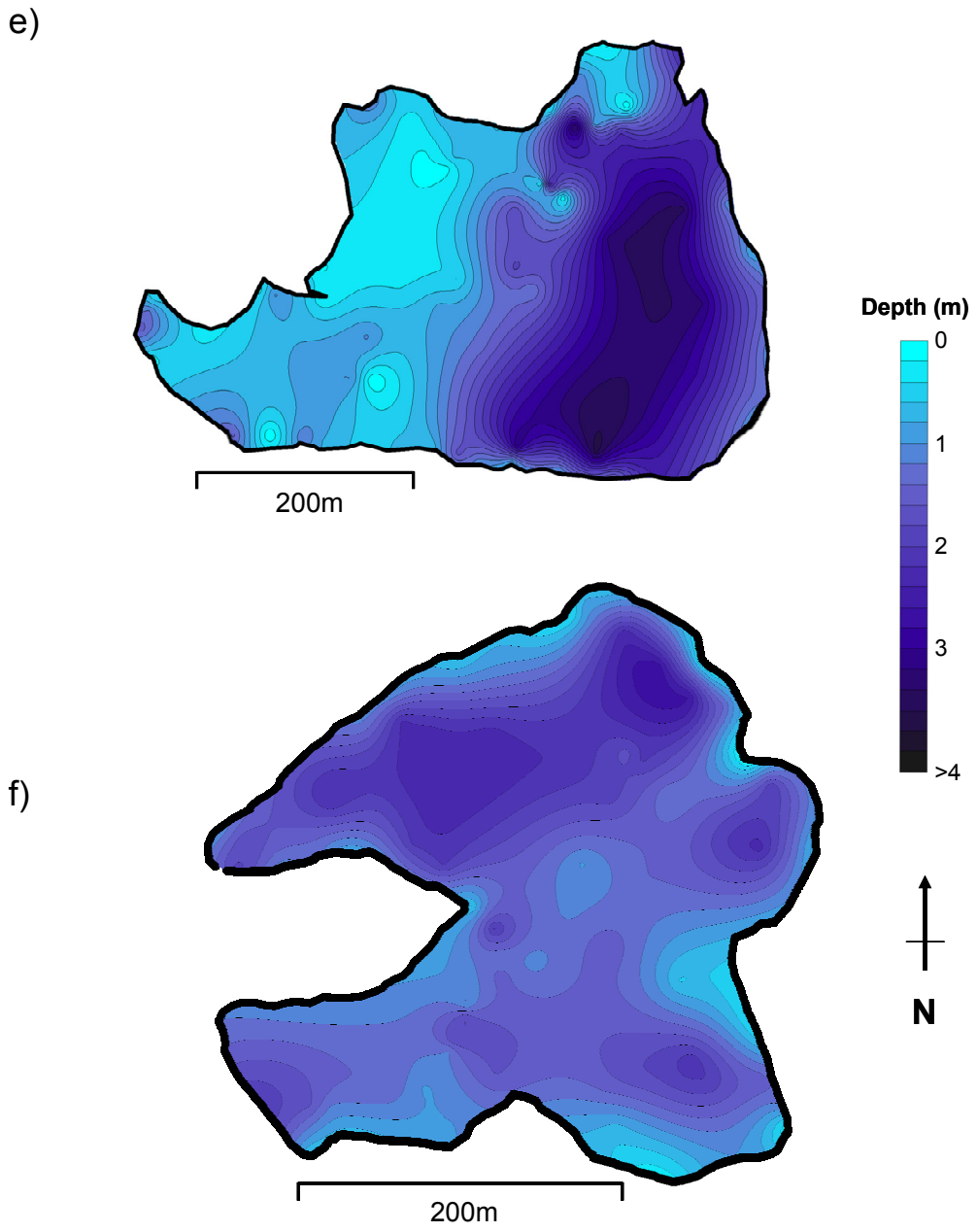


Figure 2.2contd.

have historically been high in the River Erewash. Diffuse pollution has largely been derived from mine water although improved management practices implemented in the 1990s have reduced this loading (Environment Agency, 1995). Population increases in the suburbs of Nottingham resulted in many sewage treatment works (STWs) being established along the River Erewash. Currently, seven STWs discharge directly into the River Erewash whilst one discharges into the Giltbrook

tributary. Improvements in the processing abilities and capacity of STWs driven by the 1961 Rivers (Prevention of Pollution) Act helped to mitigate against the effects of point source effluent discharge, although STW effluent still dominates the discharge of the River Erewash; during periods of low flow, up to 70% of discharge originates from STWs (Environment Agency, 1995).

The lakes of Attenborough Nature Reserve have formed part of a highly managed Nature Reserve since 1964, encompassing around 365 ha of open water, wetland, mixed woodland and grassland. Part of the Nature Reserve is designated a Site of Special Scientific Interest (SSSI) due to the importance of the site for overwintering wildfowl, particularly pochard (*Aythya ferina*) and shoveler duck (*Anas clypeata*), and the variety of terrestrial and aquatic habitats (Natural England, 1981). A number of other locally and nationally rare bird species have been recorded around the reserve, including red kite (*Milvus milvus*), carageny (*Anas querquedula*), kingfisher (*Alcedo atthis*) and bittern (*Botaurus stellaris*). Many areas of the reserve are managed in order to encourage wildfowl. For example, areas of mudflats have been created by remodelling some shorelines (Figure 2.4) and reedbeds have been established adjacent to the visitor centre and Clifton Pond (Figure 2.5). The reserve is a popular visitors attraction for educational and recreational purposes. A state-of-the-art visitors centre (Figure 2.6) was opened in 2005 to encourage tourism and provide educational resources. The lakes are also popular with anglers and used by a local sailing club.

Figure 2.3 Barges transport extracted aggregates through the lakes to the aggregate processing plant, on the northern edge of Main Pond.

Figure 2.4 Areas of mudflats created in Lake C2.

Figure 2.5 Reedbeds established along the western shoreline of Lake I2.

Figure 2.6 The visitor centre at the Attenborough Nature Reserve.

The Attenborough Ponds have been subject to numerous alterations to their morphology before reaching their current configuration. Before 1972, the River Erewash discharged directly into the River Trent. On the completion of aggregate extraction, the River Erewash was diverted into the inter-connected lakes C1, C2 and C3 to allow the passage of barges associated with extraction activity beyond the bounds of the Nature Reserve. The weir along the eastern edge of Lake C3 (MPO) became the dominant outflow. As an attempt to reduce the negative impacts of nutrient-rich River Erewash water on the Attenborough Ponds, the weir at MPO was raised in 1980 causing Ea to become the dominant outflow (Figure 2.1). The last significant alteration to the lakes occurred in 1981, when Lake I2 was isolated from all other lakes by the construction of earth bunds (Sayer and Roberts, 2001). Lakes I1 and I2 have no major inflows or outflows, although small quantities of water have been observed flowing out through a small pipe in the northern end of Lake I2 and into the River Trent. Lake I3 is fed by a small inflow (BPI) along the north-western boundary, and discharges through the Delta Sanctuary and into the River Trent. The last significant alteration was the diversion of the River Erewash to flow directly into the River Trent during August 2009.

2.2 Previous research

Previous research on the Attenborough Ponds has highlighted the extent of the changes in ecology and chemistry that occurred after the 1972 diversion of the River Erewash into the lakes. This has been both contemporary (Petts et al., 1995) and palaeoecological (Sayer and Roberts, 2001). The most significant changes in the Erewash-connected lakes were a decline in submerged (including *Potamogeton* spp., *Myriophyllum* spp., and *Elodea* spp.), floating-leaved (including *Nuphar lutea* and *Polygonum amphibium*) and marginal (including *Typha latifolia* and *Iris pseudacorus*) macrophyte communities by around 1988 (Petts et al., 1995). A decline in the vegetation abundance of Lake I3 occurred prior to 1976, but this is believed to have been a consequence of storm sewage overflow, rather than connectivity to the River Erewash. Macroinvertebrate populations generally declined in diversity after the River Erewash diversion. The abundance of some pollution-tolerant species increased after the diversion. In Lake I3, for example, *Tubifex* and Chironomidae increased and *Gammarus tigrinus* increased in abundance and range throughout the lakes. However,

an accurate assessment of the impact of the Erewash diversion on the macroinvertebrate community is difficult since the historical data available are sparse and often qualitative (Petts et al., 1995).

Sayer and Roberts (2001) used palaeoecological evidence to investigate the effect of the 1972 Erewash diversion on the diatom community of the Attenborough Ponds and to suggest which of the isolated lakes represents a suitable analogue for a restoration target. In Lake C2 after the diversion of the Erewash, a dramatic shift in diatom production occurred, from abundant benthic (particularly *Gyrosigma* spp.) and periphytic species towards dominance of planktonic species, suggesting a decline in light climate and a loss of submerged macrophytes (Sayer and Roberts, 2001). Although diatom species indicative of eutrophic conditions were found in pre-diversion sediments, the presence of the typically mesotrophic diatom *Cyclotella radiosa* in those sediments suggests much lower nutrient concentrations pre-diversion. This corresponded well with nutrient chemistry data gathered pre-diversion (Figure 2.7).

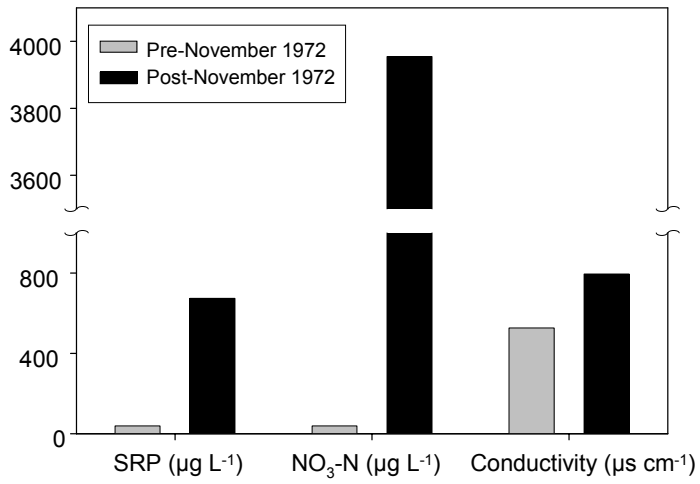


Figure 2.7 Summary of water chemistry data pre- and post- 1972 in Lake C2. Pre diversion mean derived from three samples collected between September and November 1972. Post diversion means calculated from one year of monthly monitoring data subsequent to November 1972 (data from Britton, 1974, in Sayer and Roberts, 2001).

2.3 Rationale for site choice

Previous research has shown the extent of biological and chemical change within the Attenborough Ponds as a result of the diversion of the River Erewash in 1972 (section 2.2). As has already been illustrated in Chapter 1, there is a clear need to quantify the effects of nutrients and hydrology on shallow lakes. This is particularly the case for lowland areas where anthropogenic pressures are typically greater than in upland areas. Lowlands are often subject to more diffuse nutrient sources from arable agriculture, and more point-source nutrient loading associated with urbanisation and STWs (see section 1.4, page 34).

The current geography of the Attenborough Ponds allows the nutrient-rich water from the River Erewash to penetrate only some of the lakes in the system. The result of this is that three lakes are hydrologically connected to the River Erewash and three are currently isolated (see Figure 2.1, page 74). This provides a good opportunity to quantify the effects of nutrient-rich inflows on shallow lowland lakes, in effect creating a semi-experimental system in which external influences (for example climate and geology) are controlled, whilst connectivity to the River Erewash varies.

Cemex U.K. Ltd. are implementing a strategy to reduce the nutrient loading on the Attenborough Ponds which aims to improve water quality in the lakes connected to the River Erewash (HR Wallingford, 2006; White Young & Green, 2006). The River Erewash will be diverted from the connected lakes to restore its original confluence with the River Trent. The scheme is to be implemented by the construction of an earth embankment at the current junction between the River Erewash and Lake C1, such that the connected lakes will receive <1% of current annual inflow from the River Erewash (White Young & Green, 2006). A gap remains in order to allow the passage of barges (Figure 2.8). The scheme became operational in August 2009. Predicting the outcome of significant changes to the hydrology and nutrient loadings of the connected lakes is of clear management interest.

Figure 2.8 The diversion of the River Erewash from Attenborough Nature Reserve using earth bunds (left and right of the photograph). A gap remains to allow the passage of barges into the Erewash-connected lakes. Photograph courtesy of C. Pointer.

2.4 Sampling procedures

This section describes common field and laboratory methods used in the gathering of data. Where methods were modified for specific samples, details are given in the relevant chapter.

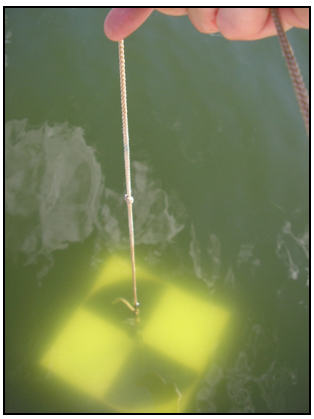
2.4.1 Water sampling and analysis

Each of the six lakes were sampled approximately every four weeks for chemical, biological and physico-chemical parameters (Figure 2.9). Lakes were sampled at points marked using anchored buoys (see Figure 2.1 for approximate locations). Sampling sites were chosen to be at the deepest point of the lake that could be sampled practically. Care was taken to position sampling sites away from the passage of barges. Data collected prior to October 2005 was provided courtesy of S. McGowan alone.

At lake sites, a ~4 L water sample from the top 1.5 m of the water column was obtained using an flexible polythene sampling tube and pooled. Inflow and outflow

sites were sampled by filling a sample bottle with surface water. Water for chemistry analysis was placed into an acid-washed Pyrex bottle. For lake samples, 1 L of pooled water was put into a plastic bottle for total suspended solids analysis (TSS). Samples were kept covered and cool until their return to the laboratory within 6 hours, where approximately half of the sample was filtered through Whatman GF/C filter papers using a vacuum pump and stored in a Pyrex bottle. Filtered and unfiltered water samples were stored in the dark at 4˚C before chemical analysis within 24 hours.

a)



b)

c)

Figure 2.9 Sampling at Attenborough Nature Reserve. a) measuring Secchi disk depth in Lake C1 during August 2006; b) flow gauging at site Ea; c) sampling Lake I2, July 2006; d) (overleaf), sampling during April 2007 (circled). Photographs a and c courtesy of S. McGowan, b, courtesy of T. Needham, d, from Google Earth (accessed 21 August 2009).

d)

Figure 2.9 contd.

Total phosphorus (TP) analysis was performed on unfiltered water. Prior to June 2007, TP samples were analysed by digestion in a pressure cooker at 1.5 psi with sulphuric acid and potassium persulphate, then concentrations were determined as described for the microwave method. Soluble reactive phosphorus (SRP) was analysed spectrophotometrically at 885 nm against a reagent blank using filtered water and the molybdate blue and ascorbic acid method (Mackereth et al., 1978). From June 2007, TP was analysed using a microwave digestion method. Samples for TP were heated to 120°C at 3.5 psi for 40 minutes in an Anton Paar Multiwave 3000 microwave digester, with a potassium persulphate and sodium hydroxide oxidising solution. Concentrations were derived after colourimetric analysis against a reagent blank at 885 nm as ortho-phosphate using the molybdate blue and ascorbic acid method (Mackereth et al., 1978).

Ammonium (NH_4-N) analysis was performed on filtered samples. Samples were reacted in alkaline conditions with sodium nitroprusside, forming indophenol blue which was measured spectrophotometrically at 640 nm (Mackereth et al., 1978) and converted to concentrations (µg L^{-1}) using a calibration curve prepared simultaneously. Nitrate (NO_3-N) was determined using an ion chromatography method described below (page 81).

Unfiltered samples were analysed for silicate (SiO_3) using the molybdate yellow reactive method and spectrophotometric measurement at 365 nm (Eaton et al., 1995). The molybdate yellow method quantifies only Si which is molybdate-reactive, which is the fraction of Si that is cycled through biological processes. Unreactive forms of silica may be present in filtered samples in addition to molybdate-reactive forms, contributing to the total quantity of dissolved Si present (Eaton et al., 1995). Reactive Si, however, forms by far the dominant dissolved fraction in natural waters (Burton et al., 1970).

pH, specific conductivity (mS cm^{-1}), temperature (°C) and dissolved oxygen (DO, mg L^{-1}) were was determined in the field using a YSI 600QS-O-M multiprobe attached to a 650MDS data logger, at 50 cm depth intervals from the surface of the lake to sediment surface.

Cations (magnesium, Mg^{2+}, calcium, Ca^{2+}, sodium, Na^{+}; and potassium, K^{+}) were determined using a Metrohm Basic 792 ion chromatography system using filtered samples. Metrosep C2-150 column was used with a 4.0 mmol L^{-1} tartaric acid and 0.75 mmol L^{-1} dipicolonic acid eluent with a flow rate of 1.0 mL min^{-1}. The anions Chloride (Cl^{-}) and NO_3-N were measured using a Metrosep A Supp 4-250 column and eluent containing 1 mmol sodium bicarbonate and 3.2 mmol of sodium carbonate run at 1.0 mL min^{-1}. Chromatogram peaks were calibrated using pre-determined calibration curves. Total alkalinity (meq L^{-1}) was determined by titration to grey colour using phenolapthalein and BDH indicators using unfiltered water samples, and total alkalinity defined as the sum of both titrations (Golterman et al., 1978).

Total suspended solids were determined by filtering a known volume of agitated water sample using a vacuum pump through pre-weighed Watman GF/C glass fibre filter papers. After drying at 105°C for at least 15 hours, the filter papers were re-weighed and the weight of suspended solids calculated as mg L^{-1} using $((W_2 - W_1) \times 1000) / V$ where W_1 was the weight after filtering (g), W_2 weight before filtering (g) and V the volume of water filtered (mL). Light penetration (cm) was measured using the mean value of one descending and one ascending Secchi disk depth.

2.4.2 Phytoplankton sampling and analysis

Total algal biomass (measured as chlorophyll-*a*) was estimated by filtering a known quantity of lake water through 47mm Watman GF/C glass-fibre filter papers. Filter papers were immediately wrapped in labelled foil, then cut into strips and placed in glass vials. After the addition of ~5 mL of extraction solvent (comprising 80% acetone, 15% methanol and 5% distilled water), samples were placed in a refrigerator at 4°C for 12 hours. The solvent was then decanted into centrifuge tubes and centrifuged at 700g for 10 minutes. Samples were then analysed spectrophotometrically against an extraction solvent blank according to the equations of Jeffrey and Humphrey (1975). Chlorophyll-*a* data relating to the period March 2005 to October 2005 was provided by S. McGowan,

15 mL of the pooled water chemistry sample was placed into a centrifuge tube for phytoplankton identification. This sample was preserved immediately using Lugol's iodine. Phytoplankton were quantified and identified using the Untermöhl (1958) technique. A known volume of water was placed in a sedimentation cell and left for at least 12 hours. Phytoplankton were identified to at least genus level at 400x magnification under a Zeiss Axiovert 40c inverted microscope, using Prescott (1954), Bellinger (1992) and John et al. (2002). A minimum of 300 phytoplankters were counted in each sample. Diatoms were identified to genus level, except for small centric types (e.g. *Cyclotella* spp. and *Stephanodiscus* spp.). *Aulacoseira* spp. were identified separately because of their filamentous form. Cell biovolume was estimated by measurement and geometric approximation of cell shape to 3-dimensional shapes (e.g. Wetzel and Likens, 1991; Hillebrand et al., 1999). The biovolume estimates of centric diatoms were based on small cells (diameter <5 µm) and large cells (>5 µm). Biovolume estimates and a species list are given in Appendix 1. Phytoplankton samples were collected by S. McGowan prior to October 2005, although all phytoplankton samples were analysed by the author.

2.4.3 Zooplankton sampling

Quantitative estimates of the zooplankton community were made using a custom-built device, comprising a weighted bucket of 160 × 160 mm with a removable 250 µm mesh attached to the base (Figure 2.10) approximately 5 cm in diameter. The sampler was vertically towed a known distance through the water column. Care was taken to ensure that as much of the full height of the water column was sampled as possible and that a constant speed of towing (approximately 0.5 m sec^{-1}) was used to avoid the formation of a bow wave or to allow individuals to swim free of the trap (Wetzel and Likens, 1991). After removing the mesh, zooplankton were rinsed into sampling jars using distilled water and immediately preserved using a methanol-sucrose solution. The procedure was repeated at least twice and the samples pooled to produce a representative sample of the zooplankton community. Zooplankton samples collected prior to October 2005 were provided by S. McGowan, although the identification and enumeration of all samples was done by the author.

Zooplankton were identified by subsampling a known volume of the sample, and counting and identifying at least 100 individuals to genus or species level. Where fewer than 100 individuals were present in the total samples, the whole sample was counted. The subsample was placed into a counting chamber and examined using a 4–50× magnification Wild Heerburgg binocular dissecting microscope. Where required, zooplankton were examined in further detail using a compound microscope at ×100 magnification. Identifications were made with reference to Scourfield and Harding (1966) for Cladocera, Harding and Smith (1974) for copepods and Pontin (1978) for rotifers. A species list is given in Appendix 2.

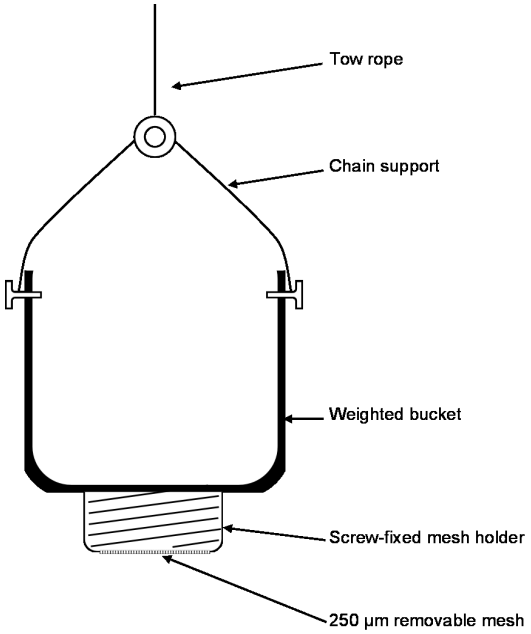


Figure 2.10 Custom built zooplankton sampler for shallow lakes.

2.4.4 Vegetation Surveys

Vegetation surveys were conducted on all lakes in 2005 (by S. McGowan), in addition the isolated lakes I1 and I2 were also surveyed during 2006 and 2007 by S. McGowan (Lake I2, 2007) and the author (all other surverys). Surveys were timed where possible to coincide with maximum vegetation abundance, between late July and early September. Submerged and emergent vegetation was identified at approximately 100 randomly chosen sites throughout each lake. The location of each site was recorded using a handheld GPS with an accuracy of ±5 m. Species were identified to at least genus level with reference to Haslam et al. (1975) using a bathyscope where possible, or samples were retrieved for more detailed analysis using a grapnel.

Chapter 3 Effects of the River Erewash on the lakes of Attenborough Nature Reserve: results and trends

3.1 Introduction

This chapter aims to compare how hydrological connectivity to the nutrient-rich River Erewash affects the structure and functioning of the lakes of Attenborough Nature Reserve. Using data gathered between March 2005 and March 2008, this is addressed by answering the following questions:

- Are there any differences in water quality and seasonal changes among connected and isolated lakes?
- Are there differences in the total biomass of phytoplankton and zooplankton among connected and isolated lakes?
- Are there differences in phytoplankton and zooplankton communities among connected and isolated lakes?

The data collected represent an opportunity to compare the structure and functioning of shallow lakes connected to a nutrient-rich river, subjected to shortened water retention times and increased nutrient loading, to those that have remained isolated. An understanding of this is essential for the management of shallow lowland lakes, given future scenarios of increasing urbanisation pressures and increased variability in river discharge resulting from climate change (IPCC, 2008). Some authors have suggested that hydrological connectivity increases the concentrations of nutrients, except P (Glińska-Lewczuk, 2009). Theoretical and empirical evidence suggests that phytoplankton biomass is higher where flushing rates are reduced, and larger species become more dominant (e.g. Dickman, 1969, Reynolds, 2006). This chapter evaluates these processes in the highly eutrophic lakes of Attenborough Nature Reserve. The data presented additionally provide an important baseline of monitoring data to assess the future effects of the diversion of the River Erewash on connected lakes.

The methods of data collection and laboratory methods are given in Chapter 2 (page 73). Means are given with the standard error shown in parentheses and the phrase 'monitoring period' refers to March 2005 to March 2008 unless specified otherwise. Significance of long-term trends in variables was assessed using Spearman's rank correlation coefficient (r_s) computed with SPSS 15.0. Only significant ($p \leq 0.05$) coefficients are reported in the text. Analysis of phytoplankton and zooplankton communities was done using CANOCO 4.5 (ter Braak and Šmilauer, 2002). Phytoplankton species were grouped to genus level, and genera that contributed less than 5% of the biovolume to a sample were excluded prior to analysis (e.g. Raikow et al., 2004) to remove noise from the dataset. Phytoplankton and zooplankton datasets were $\log_{(x+1)}$ transformed prior to analysis in order to stabilise variances (Ramette, 2007).

The presentation of data follows a bottom-up approach by beginning with water chemistry, followed by phytoplankton and zooplankton communities. This bottom-up approach is based on the theory that the availability of resources (such as nutrients) is the main determinant of the biomass of phytoplankton (e.g. Vollenweider, 1968; Dillon and Rigler, 1974; Hecky and Kilham, 1988; Elser et al., 1990). Alternative explanations suggest that phytoplankton biomass is determined by a combination of bottom-up and top-down processes, whereby the potential biomass of phytoplankton is a product of bottom-up processes and the 'realised' biomass is a result of grazing by zooplankton (McQueen et al., 1986). Experimental and observational evidence often suggests that both bottom-up and top-down effects may be important (e.g. McQueen et al., 1989; Moss et al., 1994; Stephen et al., 1998). Resolving the mechanisms controlling phytoplankton biomass is of particular relevance to improving the success of lake restoration (de Melo et al., 1992). This uncertainty is addressed in Chapter 4.

3.2 Lake C1 (Coneries Pond)

Figure 3.1 A south-westerly view across Lake C1 (Coneries Pond), September 2008. Coneries Pond is the first lake into which the River Erewash discharges. The sample buoy is in the centre-right of the photograph. Ratcliffe-on-Soar power station is in the background. Attenborough Sailing Club can be seen to the right of the image.

3.2.1 Water chemistry

The mean total phosphorus (TP) concentration in Lake C1 (Figure 3.1) was 549 (±30) µg L^{-1} during the monitoring period (Figure 3.2a). TP concentrations rose throughout the summers of 2005 and 2006 to reach maximum concentrations during August (990 µg L^{-1} in 2005 and 1158 µg L^{-1} during 2006). In 2007, TP concentrations rose during early summer but did not exceed 620 µg L^{-1} before declining at the end of the monitoring period. Soluble reative phosphorus (SRP) concentrations followed a similar seasonal trend. The mean SRP concentration observed during the monitoring period was 356 (±23) µg L^{-1}, with the highest concentrations being observed in August 2005 (710 µg L^{-1}). Maximum SRP concentrations observed during 2006 and 2007 were 689 µg L^{-1} and 554 µg L^{-1} respectively.

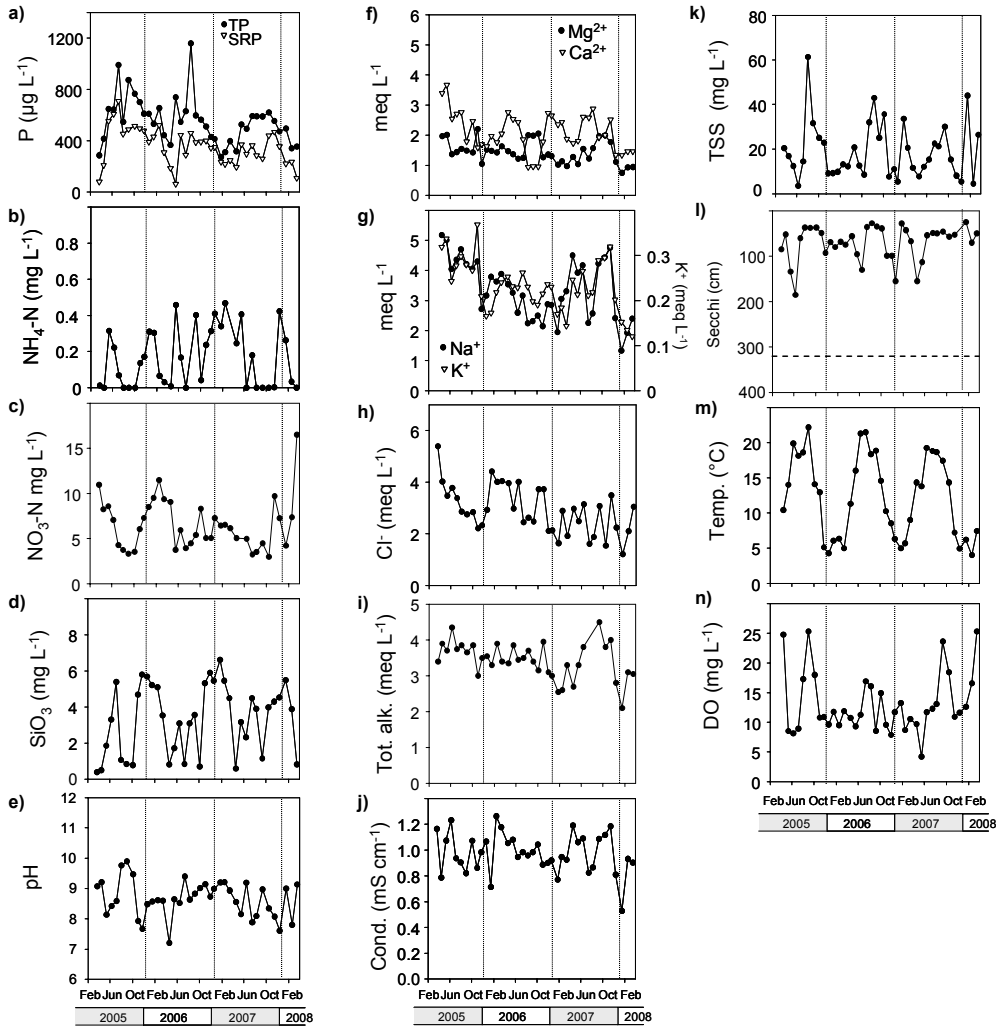


Figure 3.2 Water chemistry and physico-chemical properties of Lake C1 (Coneries Pond), March 2005 to March 2008: a) TP and SRP (µg L^{-1}); b) NH_4-N (µg L^{-1}); c) NO_3-N (mg L^{-1}); d) SiO_3 (mg L^{-1}); e) pH; f, Mg^{2+} and Ca^{2+} (meq L^{-1}); g) Na^+ and K^+ (meq L^{-1}); h) Cl^- (meq L^{-1}); i) total alkalinity (meq L^{-1}); j) specific conductivity (ms cm^{-1}); k) TSS (mg L^{-1}); l) Secchi disk depth (cm, dotted line represents approximate lake depth); m) surface temperature (°C); n) DO (mg L^{-1}).

Ammonium (NH_4-N) concentrations were highly variable throughout the monitoring period (Figure 3.2b). NH_4-N concentrations were typically highest during early summer and the winter, and were frequently below detectable concentrations during the late summer. The highest concentration recorded was 0.47 mg L^{-1} in February 2006, with a mean of 0.17 (±0.03) mg L^{-1} for the whole monitoring period. Nitrate (NO_3-N) concentrations in lake C1 ranged between 3.0 and 16.5 mg L^{-1} throughout the

monitoring period. During 2005, NO_3-N concentrations fell from 11.0 mg L^{-1} in March to 3.3 mg L^{-1} by August (Figure 3.2c). Concentrations then rose gradually throughout the winter to 11.5 mg L^{-1} by February 2006, before falling sharply to early summer. During 2007, NO_3-N did not rise as much as in 2005 and 2006, falling gradually throughout the summer from 8.3 mg L^{-1} in October 2006 to 2.9 mg L^{-1} in October 2007 and then rising sharply at the end of the monitoring period to 16.5 mg L^{-1} in March 2008.

Silicate (SiO_3) concentrations (Figure 3.2d) varied seasonally in Lake C1. The mean SiO_3 concentration was 3.3 (±0.3 mg L^{-1}). Concentrations typically rose briefly from < 1 mg L^{-1} in spring to in excess of 3 mg L^{-1} by early summer, before falling sharply during late summer and rising abruptly again during winter. The highest SiO_3 concentration was recorded during January 2007 (6.2 mg L^{-1}) and the lowest concentration (0.7 mg L^{-1}) in April 2006. Mean growth season SiO_3 concentrations increased slightly throughout the monitoring period, from 2.28 (±0.9) in 2005 to 3.00 (±0.6) mg L^{-1} in 2008.

Surface pH (Figure 3.2e) ranged from 7.20 to 9.89 throughout the monitoring period. In 2005, pH was highest during late summer before declining during the winter. pH remained relatively constant throughout 2006, and then fell during spring and early summer in 2007. pH oscillated from summer 2006 to the end of the monitoring period. Growth season mean pH fell each year, from 8.96 in 2005, to 8.52 in 2007. Profiles of pH throughout the water column (Figure 3.3a) show that throughout the monitoring period pH did not change substantially from the surface down the water column.

Magnesium (Mg^{2+}) concentrations (Figure 3.2f) in Lake C1 ranged between 0.75 and 2.20 meq L^{-1}, with a mean of 1.45 (±0.06) meq L^{-1} throughout the monitoring period. A trend towards lower concentrations occurred throughout the monitoring period (r_s = -0.399, $p \leq 0.013$). Highest concentrations tended to occur during winter months and lowest concentrations during the summer, although during 2007 Mg^{2+} concentrations rose gradually throughout the summer, before falling to the end of the monitoring period. The mean calcium (Ca^{2+}) concentration in Lake C1 was 2.10 (±0.10) meq L^{-1}, ranging between 0.94 and 3.67 meq L^{-1}. Ca^{2+} concentrations (Figure 3.2f) declined throughout 2005 and rose again from December 2005 to April 2006, before declining

sharply during summer 2006. During 2007, Ca^{2+} concentrations did not decrease throughout the summer to the same extent as previous years, although they fell gradually from August 2008 to the end of the monitoring period.

Sodium (Na^+) and potassium (K^+) concentrations showed similar trends in concentrations (Figure 3.2g) both with a significant declining trend throughout the monitoring period (Na^+ r_s = -0.448, p = 0.005; K^+ r_s = -0.436, p = 0.006). Na^+ concentrations ranged from 1.33 meq L^{-1} to 5.17 meq L^{-1} (mean 3.35 (±0.16) meq L^{-1}) and K^+ ranged from 0.12 meq L^{-1} to 0.37 meq L^{-1} with a mean of 0.23 (±0.01) meq L^{-1}. Both Na^+ and K^+ concentrations generally decreased from March 2005 to late winter 2006, before rising until November 2007. Concentrations of both Na^+ and K^+ fell sharply until the end of the monitoring period. The mean chloride (Cl^-) concentration was 2.85 (±0.13) meq L^{-1} during the monitoring period. A significant decreasing trend in Cl^- concentrations was observed (r_s = -0.509, p = 0.001). Cl^- concentrations (Figure 3.2h) declined sharply from March 2005 until November 2005 and then increased substantially until January 2006. Beyond May 2006, Cl^- concentrations became more variable, gradually decreasing towards the end of the monitoring period.

Total alkalinity (Figure 3.2j) declined until early 2007. Prolonged increases in total alkalinity occurred during 2007, followed by a sharp decrease to March 2008. Mean total alkalinity was 3.44 (±0.08) meq L^{-1} and ranged from 2.10 to 4.50 meq L^{-1} throughout the monitoring period. Phenolphthalein alkalinity (Figure 3.2i) ranged between 0 and 1.11 meq L^{-1} (mean 0.46 ± 0.07 meq L^{-1}). In all years, phenolphthalein alkalinity was highest during late summer, declined sharply and remained at 0 during winter months before increasing throughout summer months.

Specific conductivity in Lake C1 ranged from 0.53 to 1.26 mS cm^{-1}, with a mean of 0.62 (±0.01) mS cm^{-1} throughout the monitoring period. Specific conductivity did not vary seasonally (Figure 3.2j). During 2005, specific conductivity reached a maximum in June and fell during summer, and increased sharply from January to February 2006. Specific conductivity declined gradually throughout 2006, and increased again from January to April 2007. Specific conductivity decreased in summer 2007 increased during the autumn and declined sharply from November 2007 to January 2008. Water column profiles (Figure 3.3a) show homogeneity of specific conductivity

throughout the depth of Lake C1 during the monitoring period. Only in July 2006 and March 2007 did any stratification occur, although on both occasions only a small increase in specific conductivity was observed below 2 m.

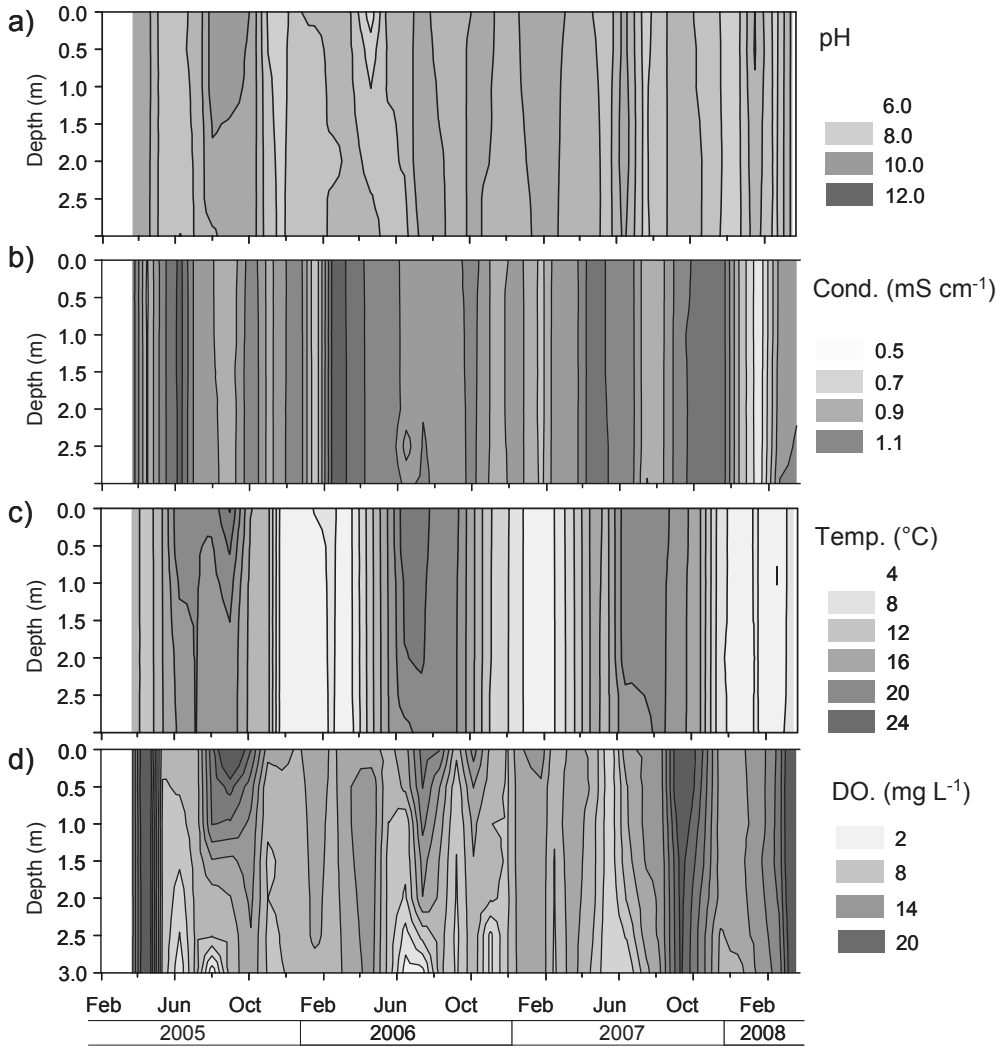


Figure 3.3 Profiles of a) pH; b) specific conductivity (mS cm^{-1}); c) temperature (°C); d) dissolved oxygen (mg L^{-1}) in Lake C1 (Coneries Pond), March 2005 to March 2008.

Total suspended solids (TSS) concentrations (Figure 3.2k) ranged from 3.4 to 61.3 mg L^{-1} (mean, 19.0 ±2.0 mg L^{-1}) during the monitoring period. During 2005 and 2006, TSS concentrations were typically highest during late summer. TSS concentrations decreased from 20.4 to 3.4 mg L^{-1} from March to June 2005 before rising sharply to 61.3 mg L^{-1} by August 2005 and decreased steadily throughout the winter. Concentrations rose briefly in spring (reaching 20.8 mg L^{-1} in April 2006) and reached

a maximum of 40.3 mg L^{-1} in August 2006. A decrease in TSS concentrations during the autumn was followed by a brief increase in TSS concentrations (> 20 mg L^{-1}) during January and February 2007. TSS concentrations increased gradually throughout 2007, reaching 30.0 mg L^{-1} in September 2007, and then decreased again to December. High TSS concentrations were observed in January 2008 (44.0 mg L^{-1}) and again in March 2008 (26.4 mg L^{-1}).

Secchi disk depths (Figure 3.2l) were generally low throughout the monitoring period, associated with chlorophyll-*a* during summer and minerogenic turbidity during winter. The clearest water occurred in June 2005 (Secchi depth, 185 cm), June 2006 (130 cm) and April 2007 (155 cm). Clear water also occurred during December 2006 (Secchi disk depth, 155 cm). The mean Secchi disk depth in Lake C1 was 71 (±6) cm, ranging between 25 and 185 cm throughout the monitoring period.

Surface water temperatures (Figure 3.2m) ranged between 4.0 and 22.2°C, with a mean of 12.3 (±0.9) °C during the monitoring period. A trend towards lower maximum temperatures occurring approximately one month earlier each year was observed during the monitoring period. Lowest temperatures each season were recorded during December and January and were always < 5°C. Temperature profiles (Figure 3.3c) showed that stratification was weak and only occurred during the summer months of each year, otherwise remaining largely homogenous.

The mean dissolved oxygen (DO) concentration (Figure 3.2n) in Lake C1 was 13.3 (±0.9) mg L^{-1}, ranging between 4.2 and 28.1 mg L^{-1}. DO concentrations were highest during late summer and early autumn. The highest DO concentrations observed during 2006 (16.9 mg L^{-1}) were lower than in 2005 (24.7 mg L^{-1}) and 2007 (23.6 mg L^{-1}). DO concentrations also rose sharply from January to March 2008. Weak stratification occurred during summer 2005 and 2006, although not during 2007 (Figure 3.3d).

3.2.2 Phytoplankton

Total biomass

The mean chlorophyll-*a* concentration in Lake C1 was 73 (±13) µg L^{-1}, and ranged between 2 and 287 µg L^{-1}. Concentrations typically increased from June to reach highest concentrations during late summer (August in 2005 (287 µg L^{-1}) and 2006 (218 µg L^{-1}) , and September in 2007 (143 µg L^{-1}), Figure 3.4). Maximum concentrations decreased in consecutive summers. Concentrations decreased rapidly from summertime maxima. Smaller increases in chlorophyll-*a* concentration occurred during March and April each year, typically of ~100 µg L^{-1}, before rapid collapses, generally occurring between April and June.

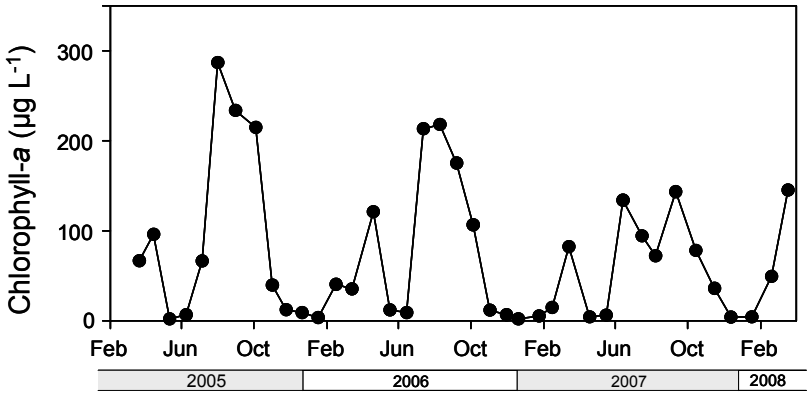


Figure 3.4 Chlorophyll-*a* concentrations in Lake C1 (Coneries Pond), March 2005 to March 2008.

Community composition

Diatoms and chlorophytes were the most abundant phytoplankton groups in Lake C1 during summer months (Figure 3.5). Diatoms were also abundant during the spring. In summer 2007, diatoms were the most abundant phytoplankton group, although in 2005 and 2006 chlorophytes were more abundant than diatoms. Cyanobacteria only formed a significant component of the phytoplankton community during summer. Cryptophytes, dinoflagellates and chrysophytes were all observed in Lake C1, although they formed a small proportion of the phytoplankton community compared to diatoms and chlorophytes.

Centric diatoms (including *Stephanodiscus* spp. and *Cyclotella* spp.) were present in the phytoplankton community throughout the monitoring period (Figure 3.6a). Centric taxa were most abundant during summer and autumn in all years. The highest biovolume of centric taxa was recorded during August 2005. A large increase in the biovolume of centric species was observed in March 2008, substantially greater than in previous years. Pennate diatoms (Figure 3.6b) were less abundant than centric taxa. During 2005 and 2006, the maximum biovolume of pennate species occurred in August. Throughout most of 2007, pennate species were present in very low biovolumes, and were only observed in early summer and autumn. *Asterionella* sp.

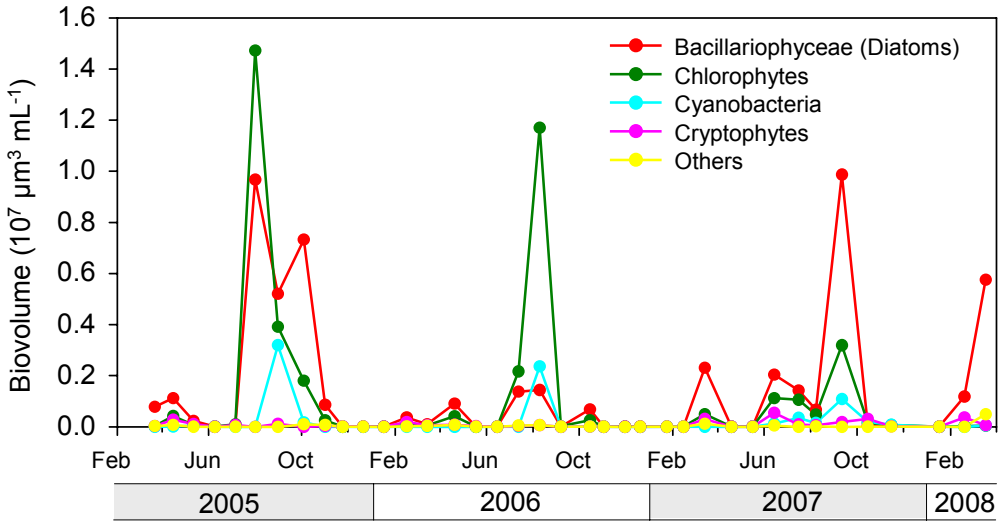


Figure 3.5 Summary of total phytoplankton group biovolumes in Lake C1 (Coneries Pond), March 2005 to March 2008. 'Bacillariophyceae' refers to all diatoms, and 'others' comprises dinophytes and chrysophytes.

(Figure 3.6c) was largely absent throughout the monitoring period, only becoming significant in the diatom community in February 2008. *Aulacoseira* spp. (Figure 3.6d) were a significant contributor to the biovolume of diatoms during summer and autumn 2005, although they formed less than 50% of the total diatom biovolume. *Aulacoseira* spp. were rare throughout 2006, and present throughout the summer of 2007, reaching 1.4 × 10^6 µm^3 mL^{-1} in September 2007. *Synedra* spp. (Figure 3.6e) were present throughout the monitoring period although were a small contributor to the total diatom biomass. During 2005, *Synedra* spp. were present during late summer and autumn, and in 2006 were most abundant in April. *Synedra* spp. were rare throughout 2007, and in March 2008 the highest biovolume of *Synedra* spp. in Lake C1 was recorded, of 9.9 × 10^4 µm^3 mL^{-1}.

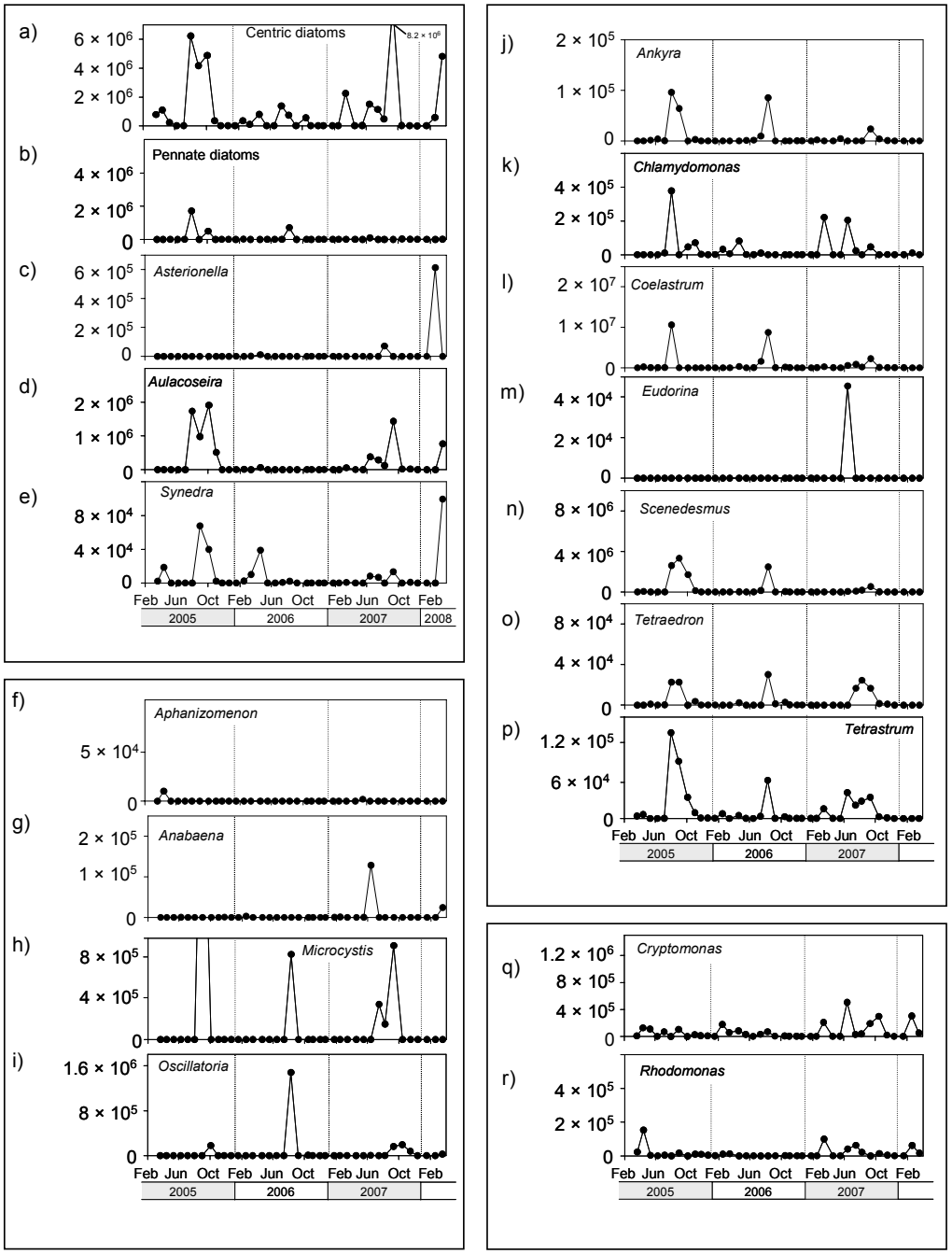


Figure 3.6 Timeseries of biovolumes (µm^3 mL^{-1}) of selected phytoplankton species in Lake C1 (Coneries Pond), March 2005 to March 2008: a-e) Bacillariophyceae (diatoms); f-i) Cyanobacteria; j-p) Chlorophyceae; q-r) Cryptophyceae. Note the differences in *y*-axis scales.

Aphanizomenon flos-aque (Figure 3.6f) was rare in Lake C1, and was only observed during April 2005 and May 2007. *Anabaena* sp. was also rare in the phytoplankton community, observed in significant biovolumes only in June 2006 (Figure 3.6g). *Microcystis* sp. (Figure 3.6h) was abundant in the cyanobacteria throughout the

monitoring period, observed during August in 2005 and 2006, and from July to September in 2007. Although *Oscillatoria* sp. was observed in late summer and autumn in all years, it was substantially more abundant in 2006 than in 2005 and 2007 (Figure 3.6i).

Chlorophytes were present throughout the monitoring period in Lake C1 and formed a large proportion of the phytoplankton community. *Ankyra judyaii* (Figure 3.6j) was present from May to August in 2005 and 2006. During 2007, *A. judyaii* was observed in May, September and October in smaller biovolumes than previous years. *Chlamydomonas* spp. (Figure 3.6k) were present in summer and autumn and again in spring and early summer of 2006, and during 2007 *Chlamydomonas* spp. were observed during early spring, summer and autumn. *Coelastrum* sp. (Figure 3.6l) was a significant contributor to the total biomass of chlorophytes in August 2005 and 2006 (in excess of 8×10^6 μm^3 L^{-1} in both years), although in 2007 the biovolume of *Coelastrum* sp. was substantially less (2.2×10^6 μm^3 mL^{-1}) and occurred in September. *E. elegans* was only recorded during June 2007 (Figure 3.6m). Members of the genus *Scenedesmus* (Figure 3.6n) formed a significant component of the chlorophyte community, particularly during summer 2005 and 2006 when the total biovolume for the genus exceeded 2.4×10^6 μm^3 L^{-1}. During 2007, the biovolume of *Scenedesmus* spp. did not exceed 0.6×10^6 μm^3 mL^{-1}. In all years, three species of *Scenedesmus* were recorded: *S. communis*, *S. falcatus* and *S. opoliensis*. All species were most frequently observed as four-celled coenobia, although two-celled coenobia were also common. Eight-celled coenobia of *S. falcatus* were observed occasionally. *Tetraedron* spp. (Figure 3.6o) were recorded throughout the summer of all years during the monitoring period although did not contribute substantially to the total biovolume of the chlorophyte community. *T. triangulare* was the most frequently observed member of the genus, and *T. caudatum* was also observed occasionally. *Tetrastrum* spp. (Figure 3.6p) were observed in all summers during the monitoring period and were most abundant during 2005, peaking at 1.35×10^5 μm^3 mL^{-1} in August 2005. In 2006 and 2007, *Tetrastrum* spp. did not exceed 6×10^5 μm^3 mL^{-1}. Other chlorophytes observed during the monitoring period include *Pediastrum* spp. (including *P. boryanum* and *P. duplex*) were present during summer 2005 and 2006. *Chlorella* sp. was frequently found in small biovolumes throughout the monitoring period, particularly during 2007. *Staurastrum* spp. were observed occasionally in

summer 2005 and 2006 although was not present in 2007. *Closterium* sp. was observed in small biovolumes in 2006 and 2007. During August 2005, *Closterium* sp. was a significant component of the chlorophyte community, reaching 4.0×10^5 μm^3 mL^{-1}.

Cryptophytes were present throughout the monitoring period. The biovolume of *Cryptomonas* spp. (Figure 3.6q) were generally greater than *Rhodomonas* spp. (Figure 3.6r). *Cryptomonas* spp. were generally more abundant during spring and summer during 2005 and 2006, with a decline in abundance during May and June. The biovolume of *Cryptomonas* spp. were higher during 2007 than the previous years and the largest biovolumes occurred at different times. During late summer 2007 the abundance of *Cryptomonas* spp. were also higher than in previous years. *Rhodomonas* spp. were more abundant during spring each year than other seasons. Other than a large peak in *Rhodomonas* spp. biovolume in April 2005, the abundance of *Rhodomonas* spp. were greater throughout 2007 than other years.

Peridinium spp. (data not shown) were observed during March or April in 2005, 2006 and 2007. Except for 2006, *Peridinium* spp. were not recorded in summer months, and was absent from April 2007 to the end of the monitoring period. *Gymnodinium* spp. (data not shown) were not observed throughout most of the monitoring period although were abundant during March 2008. *Mallomonas* sp. (data not shown) was a rare member of the phytoplankton community, but was abundant during March 2007. *Trachelomonas* sp. (data not shown) was observed occasionally in the phytoplankton community, although in relatively small biovolumes compared to other species.

3.2.3 Macrophytes

Submerged macrophytes were rare in Lake C1. Only *Fontinalis antipyretica* was observed, occurring at one sampling site (Figure 3.7). Filamentous algae were recorded at 15 sampling locations, mainly towards the southern edge of the lake. No floating-leaved macrophytes were found in Lake C1. Seven species of emergent macrophytes were observed. Along the north-western shore of the lake, *Acorus calamus* was the only species found, although the north-eastern and south-eastern shores had more diverse emergent macrophyte communities. *A. calamus* was found

at the majority of sampling sites, although occurred with other species, such as *Glyceria maxima* and *Typha latifolia*, along the north-eastern shore. *Phalaris arundinacea* was recorded along the south-eastern shore of the lake, and *Iris pseudacorus* was found at two sampling sites.

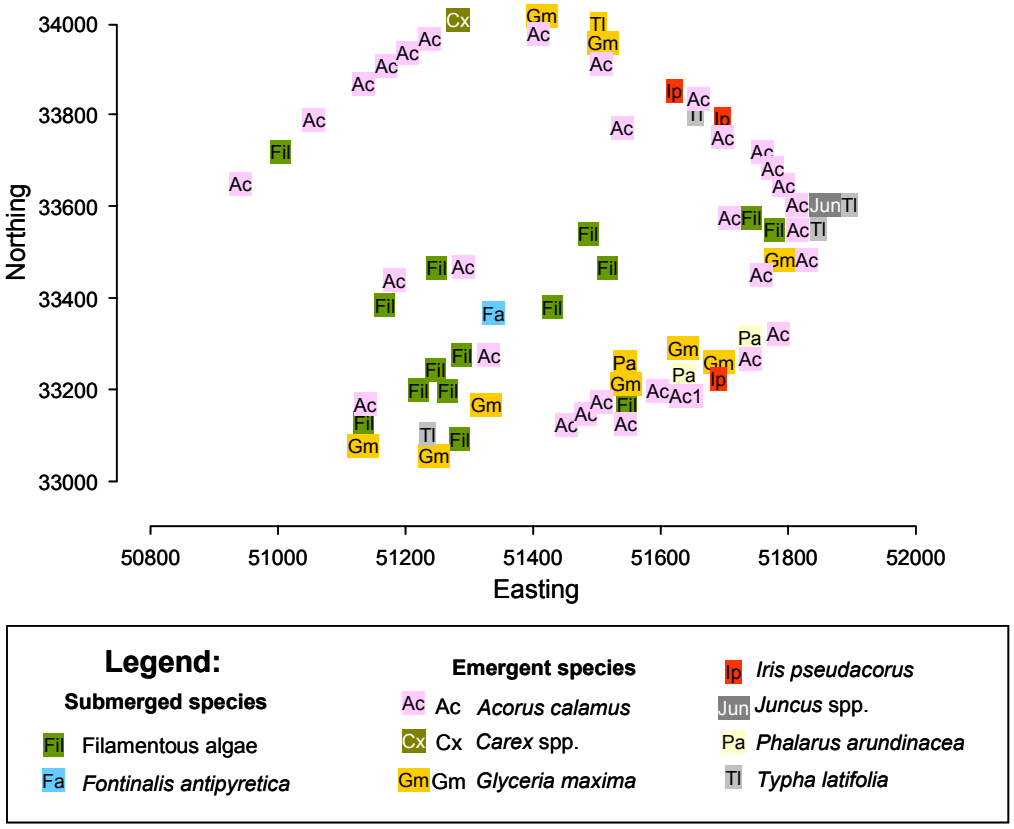


Figure 3.7 Schematic map of macrophytes observed in Lake C1 (Coneries Pond) during summer 2005.

3.2.4 Zooplankton

Total zooplankton abundance in Lake C1 (Figure 3.8a) increased steadily from 7 individuals L^{-1} in March 2005 to 32 ind. L^{-1} by October 2005. A more rapid increase in total zooplankton abundance occurred in 2006, peaking at 189 ind. L^{-1} by September 2006, before collapsing rapidly during late summer and remaining at less than 1 ind. L^{-1} until April 2007. Zooplankton abundance fluctuated throughout the summer of 2007, without the early to mid summer increases observed in the previous two years. From October 2007 until the end of the monitoring period, fewer than 2 ind. L^{-1} were observed in each sample.

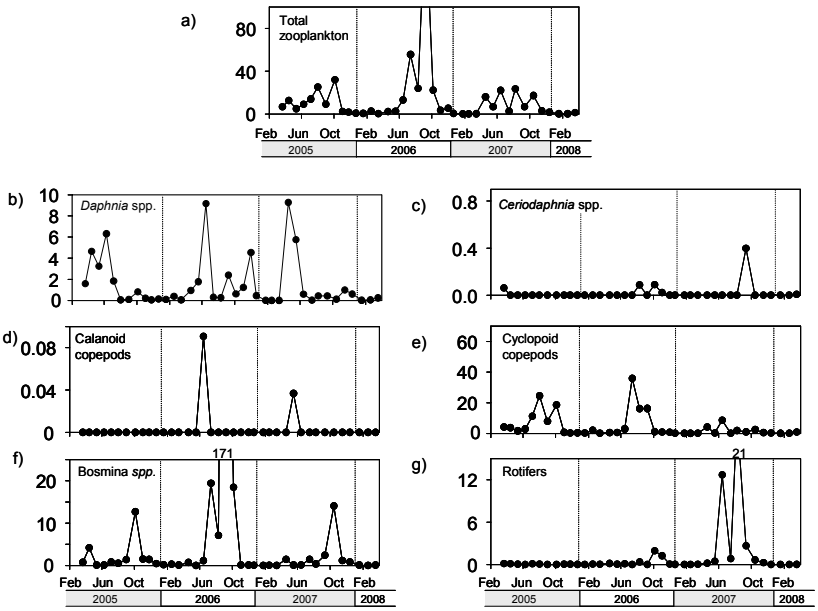


Figure 3.8 Timeseries of zooplankton abundance: a) total abundance; b-g) selected groups, March 2005-March 2008 in Lake C1 (Coneries Pond). All data are presented as individuals L^{-1}.

The abundance of *Daphnia* spp. (Figure 3.8b) was highly seasonal, peaking between May and June at around 6–8 ind. L^{-1} and collapsing to very low densities during autumn and winter. Nearly all *Daphnia* observed were *D. hyalina*, although *D. curvirostris* was observed in November and December 2008. *Ceriodaphnia* spp. and calanoid copepods were both rarely observed in the Lake C1. Cyclopoid copepods were generally the most abundant members of the zooplankton community (Figure 3.8e). Cyclopoid copepods were most abundant during late summer, in both 2005 and 2006 reaching 20–30 ind. L^{-1}. However, during 2007, cyclopoid copepods were comparatively rare with a maximum abundance of 9 ind. L^{-1} in June. *Bosmina longirostris* was also a significant contributor to the zooplankton community of Lake C1. Peak abundances of *B. longirostris* occurred during September and October in all years, reaching 12.7 and 14.0 ind. L^{-1} in 2005 and 2007 respectively. During 2006 over 170 ind. L^{-1} were recorded in September. In each year, abundances were low (<3 ind. L^{-1}) throughout the winter. Large rotifers were present in zooplankton samples throughout 2005 and 2006, generally in densities of less than 1 ind. L^{-1}, except for October 2006 when 2 ind. L^{-1} were recorded. All individuals were from the genus *Asplancha*. Rotifers were particularly abundant during August 2007 when over 20 ind. L^{-1} were observed, entirely and unusually comprising *Keratella* spp.

3.3 Lake C2 (Tween Pond)

Figure 3.9 Lake C2 (Tween Pond), photographed during September 2007. Recently constructed mud flat areas are in the foreground. Lake C2 is the second lake in the chain connected to the River Erewash. Lake C3 (Main Pond) lies to the right of the photograph and Lake C1 (Coneries Pond) to the left.

3.3.1 Water chemistry

TP concentrations in Lake C2 (Figure 3.9) ranged between 192 and 1219 µg L^{-1} with a mean of 506 (±33) µg L^{-1} during the monitoring period (Figure 3.10a). Concentrations of TP were highest during the late summer and decreased throughout the winter before increasing between spring and summer. In 2006, TP concentrations rose more rapidly than in other years, and in 2007 maximum TP concentrations were lower than in previous years (906 µg L^{-1}, 1219 µg L^{-1} and 653 µg L^{-1} in 2005, 2006 and 2007 respectively). The mean SRP concentration in Lake C2 during the monitoring period was 331 (±22) µg L^{-1}, and ranged between 49 and 661 µg L^{-1} (Figure 3.10a). SRP concentrations decreased slightly throughout monitoring period. Typically, two maxima in SRP concentrations were observed in each year, following increases in

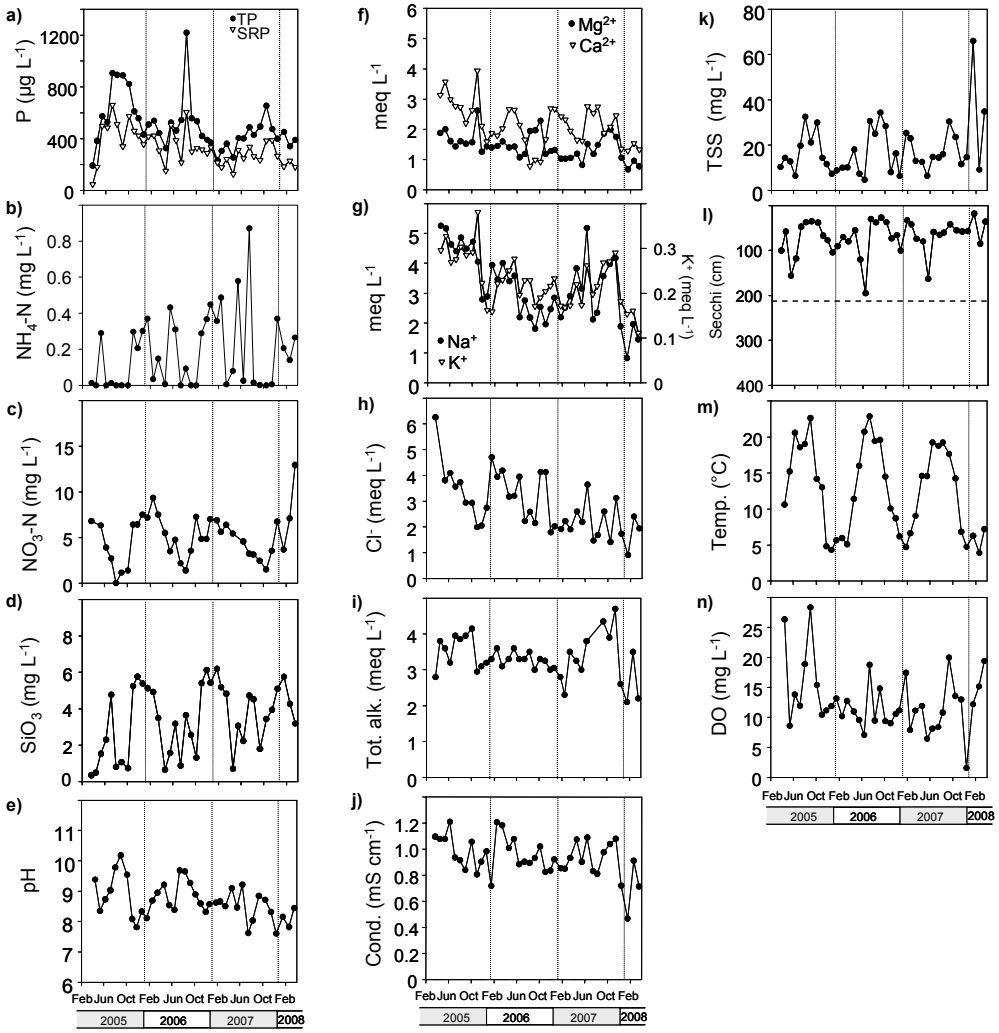


Figure 3.10 Water chemistry and physico-chemical properties of Lake C2 (Tween Pond) March 2005 to March 2008: a) TP and SRP (µg L^{-1}); b) NH_4-N (mg L^{-1}); c) NO_3-N (mg L^{-1}); d) SiO_3 (mg L^{-1}); e) pH; f) Mg^{2+} and Ca^{2+} (meq L^{-1}); g) Na^+ and K^+ (meq L^{-1}); h) Cl^- (meq L^{-1}); i) total alkalinity (meq L^{-1}); j) specific conductivity (mS cm^{-1}); k) TSS (mg L^{-1}); l) Secchi disk depth (cm, dotted line represents approximate lake depth); m) surface temperature (°C); n) DO (mg L^{-1}).

concentration during early summer and progressive decreases from autumn throughout the winter. Peaks in SRP concentrations occurred in July and October in 2005 and May and August in 2006. In 2007, this pattern was not clearly evident.

NH_4-N concentrations were highly variable throughout the monitoring period (Figure 3.10b), ranging between 0 and 0.87 mg L^{-1}. The mean concentration for the monitoring period was 0.18 (±0.03) mg L^{-1}. A trend towards higher peaks in NH_4-N concentration was apparent from March 2005 to July 2007. Typically, highest

concentrations were observed in May and June and from November to February. During summer and autumn, NH_4-N concentrations were frequently below detectable limits. In 2007, large peaks in NH_4-N concentrations were observed (0.58 mg L^{-1} and 0.87 mg L^{-1} in May and July) which decreased rapidly to August and remained low until December before rising to 0.37 mg L^{-1} and remained above 0.14 mg L^{-1} until the end of the monitoring period. The mean NO_3-N concentration was 5.0 (±0.4) mg L^{-1} throughout the monitoring period, ranging between 0 and 12.9 mg L^{-1}. NO_3-N concentrations generally decreased from concentrations of > 6 mg L^{-1} in late winter throughout spring and summer, typically to less than 2 mg L^{-1}. Then NO_3-N concentrations increased throughout late summer and autumn. A steep increase in NO_3-N concentrations was observed from October 2007 (1.5 mg L^{-1}) to March 2008 (12.9 mg L^{-1}).

SiO_3 concentrations (Figure 3.10d) were highly seasonal, ranging from 0.4 to 6.2 mg L^{-1} and had a mean of 3.4 (±0.3) during the monitoring period. SiO_3 rose between March 2005 until July 2005 after which concentrations fell abruptly and remained at ~1 mg L^{-1} until November, after which a steep increase to over 5 mg L^{-1} was observed. High concentrations were maintained over winter (> 5.1 mg L^{-1}) and fell sharply from 4.9 mg L^{-1} in February to 0.7 mg L^{-1} in April 2006. SiO_3 concentrations were variable throughout the summer of 2006, and rose sharply again to peak in November 2006 at 6.1 mg L^{-1}. In 2007, concentrations again decreased sharply during spring to 0.7 mg L^{-1} in April, and rose throughout the summer and winter to a maximum of 5.8 mg L^{-1} in January 2008, except for a sharp fall between August (4.5 mg L^{-1}) and September (1.8 mg L^{-1}) 2007.

The mean pH in Lake C2 was 8.6 (±0.1) and ranged between 7.6 and 10.2 throughout the monitoring period (Figure 3.10e). A significant decreasing trend in surface pH was observed throughout the monitoring period (r_s = -0.445, p = 0.004). During 2005 pH increased throughout the summer to reach a maximum of 10.2 in August before falling sharply during autumn to 7.8 in November 2005. pH increased gradually throughout the winter and spring, from 8.1 in January 2006 to 9.2 in April 2006. The highest pH

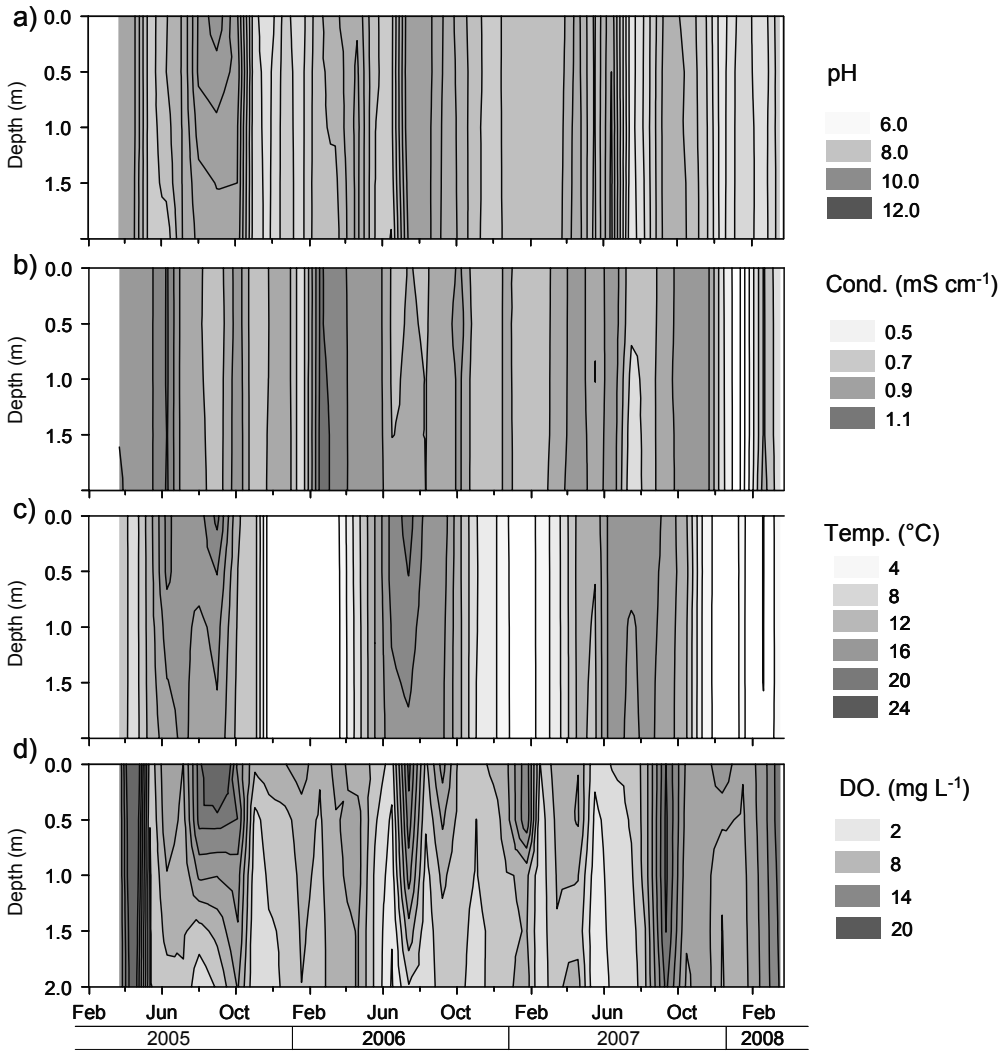


Figure 3.11 Profiles of a) pH; b) specific conductivity (mS cm^{-1}); c) temperature (°C); d) dissolved oxygen (mg L^{-1}) in Lake C2 (Tween Pond) March 2005 to March 2008.

during 2006 (9.7 in July) occurred slightly earlier than in 2005, and was followed by a more gradual decline until pH reached 8.3 in November 2006. During 2007, seasonality was not evident, although the lake was less alkaline than previous summers. Throughout 2006 and 2007, pH did not vary substantially with increasing water depth (Figure 3.11a). During August 2005, higher pH was observed nearer the surface, declining from 10.2 on the surface to 9.4 at 2 m depth.

Mg^{2+} and Ca^{2+} (Figure 3.10f) concentrations both fell gradually throughout the monitoring period (Mg^{2+}, r_s = -0.432, p = 0.007; Ca^{2+}, r_s = -0.456, p = 0.004). Mg^{2+} concentrations ranged between 0.7 and 2.6 meq L^{-1}, with a mean of 1.4 (±0.1) meq L^{-1}. The highest values of Mg^{2+} tended to occur during early summer

and November in all years. The mean concentration of Ca^{2+} was 2.1 (±0.1) meq L^{-1} and ranged between 0.7 and 2.6 meq L^{-1}. Ca^{2+} concentrations were generally highest in late summer and autumn. Na^{+} and K^{+} concentrations fell from the beginning of the monitoring period until January 2007, before increasing gradually throughout 2007 and then decreasing sharply from November 2007 until then end of the monitoring period. Throughout the monitoring period, a trend towards lower Na^{+} and K^{+} correlations was observed (Na^{+}, r_s = -0.604, $p \leq 0.001$; K^{+}, r_s = -0.536, p = 0.001). Cl^{-} concentrations (Figure 3.10h) also fell throughout the monitoring period (r_s = -0.622, $p \leq 0.001$). Seasonality was not evident, with maximum concentrations occurring in different months in each year. Mean Cl^{-} concentration was 2.8 (±0.2) meq L^{-1}, ranging between 0.9 and 4.7 meq L^{-1}. Total alkalinity (Figure 3.10i) in Lake C2 ranged between 2.1 and 4.7 meq L^{-1} with an overall mean of 3.3 (±0.1) meq L^{-1} throughout the monitoring period. Total alkalinity tended to be higher during 2007 than both 2005 and 2006.

Mean specific conductivity in Lake C2 was 0.92 (±0.02) mS cm^{-1} and ranged between 0.47 and 1.21 mS cm^{-1} during the monitoring period (Figure 3.10k). Specific conductivity showed a decreasing trend during the monitoring period (r_s = -0.416, p = 0.008) and tended to be higher during winter and summer. During 2007, specific conductivity was generally lower than in previous years. Specific conductivity fell sharply from November 2007 to January 2008, from 1.08 to 0.47 mS cm^{-1}. Specific conductivity was nearly homogenous throughout the water column during the monitoring period (Figure 3.11b).

TSS concentrations varied seasonally (Figure 3.10k). During 2005 and 2006, TSS concentrations were highest during summer, peaking in excess of 30 mg L^{-1} in each year. In both years, TSS concentrations rose rapidly from May to July and fell sharply from October to November. Late winter TSS concentrations were higher in 2006 than 2005, and reached a maximum later in the year, during September. After December 2007, TSS concentrations rose rapidly to 66 mg L^{-1} during January 2008 and decreased sharply to 9 mg L^{-1} during February 2008.

Secchi disk depths (Figure 3.10l) ranged between 18 and 195 cm, and averaged 70 (±6) cm throughout the monitoring period. Secchi disk depths were greatest (exceeding 150 cm) in May 2005 and 2007 and June 2006. Secchi depths then

decreased sharply and remained low (<65 cm) during the summer months. Lowest summer Secchi disk depths were recorded during September 2006 (27 cm). During the winters of 2005 and 2006, Secchi disk depths tended be between 50 and 100 cm, whilst during 2007 Secchi disk depths were less than 60 cm.

Surface water temperature (Figure 3.10m) was strongly seasonal throughout the monitoring period. Temperature ranged between 3.9 and 22.8°C, with a mean of 12.2 (±1.0) °C. Maximum summertime temperatures occurred between June and August in each year, exceeding 20 °C in 2005 and 2006, although in 2007 the maximum temperature recorded was 19.3 °C. Low temperatures were observed for a longer period during the winter of 2005 and 2007 than 2006. Little stratification was observed (Figure 3.11c), except during the summers of 2005 and 2006 were temperatures at the surface were approximately 4°C warmer than at 2 m depth.

DO concentrations (Figure 3.10n) were highly variable throughout the monitoring period and ranged between 1.6 and 28.4 mg L^{-1}. The mean DO concentration was 12.7 (±0.8) mg L^{-1}. DO concentrations tended to be higher during summertime, although comparable concentrations were observed during April 2005, January 2007 and March 2008. DO concentrations were generally higher nearer the surface of the lake (Figure 3.11d). Gradients of DO concentration were strongest during the summer of 2005 and 2006, when surface DO concentrations were 20.1 and 10.9 mg L^{-1} greater at the surface than 2 m depth respectively.

3.3.2 Phytoplankton

Total biomass

Chlorophyll-*a* concentrations (Figure 3.12) ranged between 2.6 and 279.0 µg L^{-1} throughout the monitoring period, with a mean of 67.0 µg L^{-1}. Chlorophyll-*a* concentrations were strongly seasonal. During 2005 and 2006, concentrations rose briefly during spring, before falling to less than 10 µg L^{-1} by May. This was followed by rapid increases in chlorophyll-*a* concentrations to maximum concentrations in August 2005 and July 2006. In both years, concentrations then decreased rapidly and remained low (<20 mg L^{-1}) until January 2006 and February 2007. Chlorophyll-*a* concentrations increased erratically throughout the summer of 2007. A smaller

summer maximum in chlorophyll-*a* concentrations was evident compared to previous summers (maximum 107 µg L^{-1}, October 2008). Concentrations of chlorophyll-*a* then decreased to 3.7 µg L^{-1} in December 2007. Chlorophyll-*a* concentrations rose from 3.7 to 43.4 µg L^{-1} from February to March 2008.

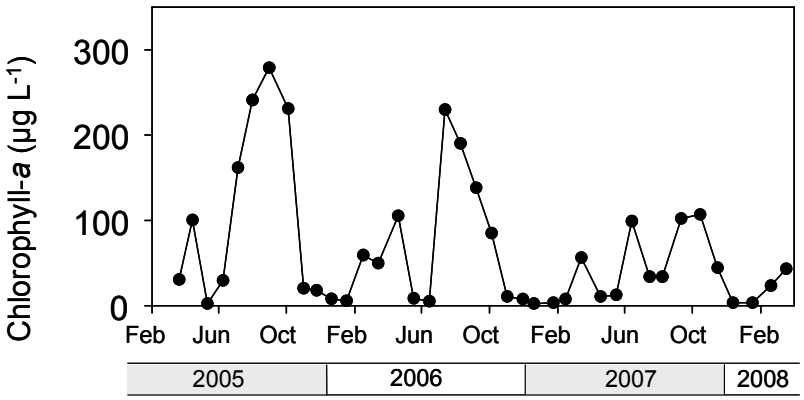


Figure 3.12 Chlorophyll-*a* concentrations in Lake C2 (Tween Pond) March 2005 to March 2008.

Community composition

Diatoms and chlorophytes were the most abundant phytoplankton groups in Lake C2 during the monitoring period (Figure 3.13). During summer 2005, diatoms were similarly abundant, although the maximum biovolume of diatoms occurred later in the year. Chlorophytes were most abundant during 2006, and diatoms were more abundant than other phytoplankton groups during 2007. Cyanobacteria were found in the phytoplankton during each year, although the highest biovolume occurred during 2006. Cryptophytes, dinophytes and chrysophytes were all found in Lake C3 although in biovolumes insignificant in comparison to other phytoplankton groups.

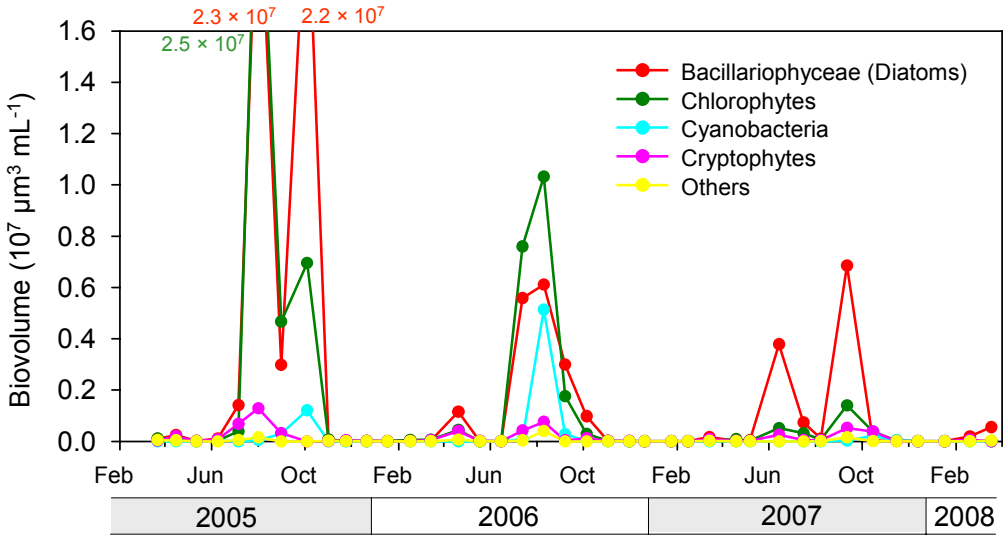


Figure 3.13 Summary of total phytoplankton group biovolumes in Lake C2 (Tween Pond) March 2005 to March 2008. 'Bacillariophyceae' refers to all diatoms, and 'others' comprises dinophytes and chrysophytes.

Diatoms were a significant contributor to total phytoplankton biomass in Lake C2 (Figure 3.14a-e). Centric diatoms (Figure 3.14a) were abundant during July to September each year, although were nearly always present in the phytoplankton community. The highest biovolume of centric diatoms was recorded during October 2005 when small centric species dominated the diatom community. Although less abundant during 2006 and 2007, centric diatoms still formed a large proportion of the phytoplankton community. Large biovolumes (> 5 × 10^6 µm^3 L^{-1}) of centric species was observed during July 2006 and September 2007. Pennate diatoms (Figure 3.14b) were not so dominant during most of the monitoring period, except during August 2006 when the biovolume of pennate species exceeded that of centric species. As with centric species, pennate diatoms were most abundant during August and September in each year. *Aulacoseira* spp. (Figure 3.14c) was recorded during summer 2005 in relatively large biovolumes reaching 8.1 × 10^6 µm^3 mL^{-1} during August 2005, and during the winters of 2005 and 2006 was present in comparatively small biovolumes. During July and August 2007 *Aulacoseira* was again abundant, although not to the extent seen in 2006. *Synedra* spp. (Figure 3.14e) were only observed in significant biovolumes during 2005 and 2006. During 2005, *Synedra* spp. were present in early and late summer and October, although during 2006 *Synedra* spp. were more abundant in March than any of the summer months. A slight increase

in the biovolume of *Synedra* spp. was observed during February and March 2008. *Asterionella* spp. (Figure 3.14d), although a rare genus throughout most of the monitoring period, became a dominant diatom briefly during February 2008, reaching a biovolume of 7.1 × 10^4 µm^3 mL^{-1}.

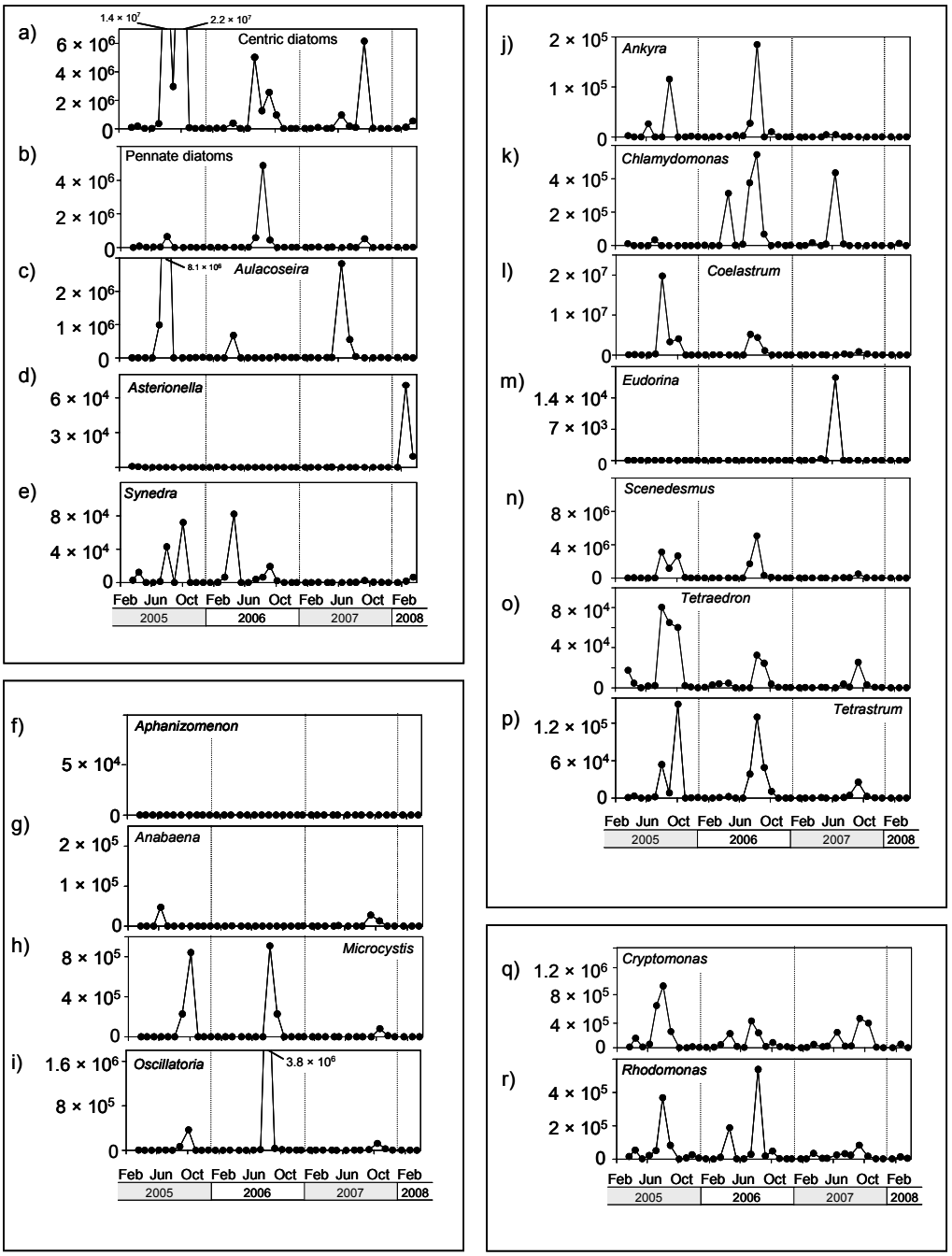


Figure 3.14 Timeseries of biovolumes (µm^3 mL^{-1}) of selected phytoplankton species in Lake C2 (Tween Pond) March 2005-March 2008: a-e) Bacillariophyceae (diatoms); f-i) cyanobacteria; j-p) Chlorophyceae; q-r) Cryptophyceae.

The cyanobacteria community (Figure 3.14 f-i) in Lake C2 comprised *Microcystis* sp. and *Oscillatoria* sp. in 2005 and 2006, with both genera being present in small biovolumes during September and October 2007. *Microcystis* sp. was most abundant during the summer of 2005 and 2006, in both years exceeding 8×10^5 μm^3 mL^{-1} during late summer. *Oscillatoria* sp. achieved a substantially higher biovolume in 2006 than 2005, and was present during the same periods as *Microcystis* sp. *Aphanizomenon* sp. was not observed during the monitoring period in Lake C2.

Chlorophytes were regularly observed in the phytoplankton community of Lake C2. All species were typically most abundant during late summer. *Ankyra judyaii* (Figure 3.14j) was present during June and August 2005, and was again abundant in August 2007, although rarely was observed throughout 2007 and 2008. *Chlamydomonas* spp. (Figure 3.14k) were an abundant member of the phytoplankton community during April 2006 and were also relatively abundant during July and August 2006. Unlike some other chlorophytes, *Chlamydomonas* spp. were also present during the summer of 2007, reaching a maximum of 4.4×10^5 μm^3 mL^{-1} in June 2007. *Coelastrum* sp. (Figure 3.14l) was abundant from July to October 2005, reaching a maximum biovolume of 1.9×10^7 μm^3 mL^{-1}, and was present in the phytoplankton community from July to September 2006, although at considerably smaller biovolumes (maximum 5.1×10^6 μm^3 mL^{-1}, July 2006). In 2007, *Coelastrum* spp. was rare, although observed in small ($< 1 \times 10^6$ μm^3 mL^{-1}) biovolumes from July to November. *Eudorina elegans* (Figure 3.14m) was only found during June 2007 in significant biovolumes. *Scenedesmus* spp. (Figure 3.14n) were the most abundant genera of chlorophyte. *S. falcatus* was the most abundant member of the genera during 2005 and 2007, whilst *S. communis* was more abundant during 2006. However, all species were observed in all years. *Scenedesmus* spp. were abundant from July to November in both 2005 and 2006, and from July to September during 2007. The highest biovolume of *Scenedesmus* spp. was recorded during August 2006 reaching 5.1×10^6 μm^3 mL^{-1}. *Tetraedron* spp. (Figure 3.14o) were more abundant during 2005 and 2006 than 2007. The highest biovolume of *Tetraedron* spp. were recorded during late summer and autumn 2005, reaching 1.5×10^5 μm^3 mL^{-1} in October. *Tetraedron* spp. were also abundant during summer 2006 (1.3×10^5 μm^3 mL^{-1}) and during the late summer in 2007. *T. caudatum*, *T. regulare* and *T. triangular* were all observed. *Tetrastrum* spp. (Figure 3.14p) were present throughout the late summer of 2005 reaching in excess of

8×10^4 µm mL^{-1} in August 2005, and present in smaller biovolumes during 2006 and 2007 (~ 3×10^4 µm mL^{-1}). *Monoraphidium* sp. was present throughout most of the monitoring period, although in low biovolumes (< 3×10^4 μm^3 mL^{-1}). *Actinastrum hantzschii* and *Ankistrodesmus* sp. were both recorded during summer 2005 and 2006. *Pediastrum boryanum*, and *P. simplex* were found in the phytoplankton occasionally, mainly during late summer in 2007. *Micratinium* sp. was observed during early summer 2007 in small biovolumes, and isolated occurrences of *Selenastrum* sp. was observed during spring and summer 2005. *Staurastrum* spp. were only observed during August 2005, in low biovolumes.

Cryptomonas spp. (Figure 3.14q) and *Rhodomonas* spp. (Figure 3.14r) were present in the phytoplankton community throughout the majority of the monitoring period. The biovolume of *Cryptomonas* spp. were generally highest at the end of summer, and were greater in 2005 compared to 2006 and 2007. Smaller increases in the biovolume of *Cryptomonas* spp. were observed during early summer. Annual maxima of *Rhodomonas* spp. biovolume occurred in August 2005 and 2006, and was greatest in August 2008. *Rhodomonas* spp. were present in substantially lower biovolumes throughout 2007 compared to previous years.

Euglenophytes (data not shown) were rarely observed in the phytoplankton community of Lake C2. *Trachelomonas* sp., *Euglena* spp. and *Phacus* spp. were all recorded sporadically, and never formed a significant portion of total phytoplankton biomass. The Dinoflagellate (data not shown) genera observed were *Gymnodinium* and *Peridinium*. *Peridinium* spp. were abundant briefly during March and August 2006. *Gymnodinium* spp. and the chrysophyte *Mallomonas* sp. (data not shown) were regularly observed although were recorded only in small biovolumes throughout the monitoring period.

3.3.3 Macrophytes

Submerged macrophytes were rare in Lake C2 (Figure 3.15). Only *Fontinalis antipyretica* was observed, and at only one sampling site. Floating-leaved macrophytes were also rare. Two species were observed (*Lemna. minor* and *Polygonum amphibium*) close to south and east shores of the lake. Seven species of

emergent macrophytes were observed. The most diverse communities were found along the eastern shores of the lake, where *Glyceria maxima* and *Acorus calamus* were most frequently observed, and *Sparganium erectum* and *Typha latifolia* were restricted to the eastern-most shore. Few emergent species were recorded along the northern and southern shores, where isolated occurrences of *G. maxima*, and *A. calamus* were recorded.

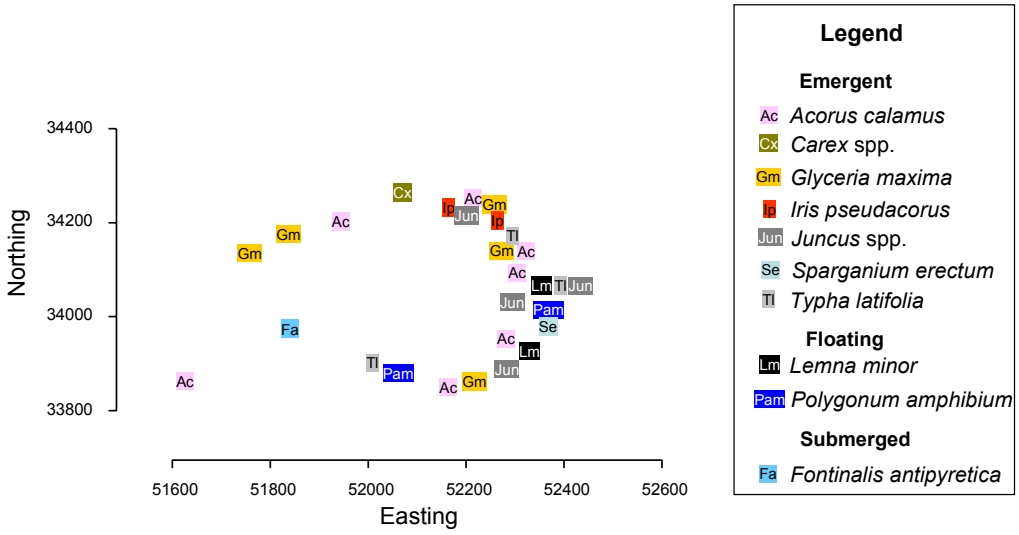


Figure 3.15 Macrophytes observed in Lake C2 (Tween Pond) during summer 2005.

3.3.4 Zooplankton

Total zooplankton abundance in Lake C2 (Figure 3.16a) was characterised by rapid increases during early summer followed by rapid decreases from late summer in 2005 and 2006. In both years, total abundance exceeded 45 ind. L^{-1}, and was slightly higher during 2005 (58 ind. L^{-1}) than 2006. During 2007, late summer increases in zooplankton abundance were of a substantially smaller magnitude than the previous two years, reaching 8 ind. L^{-1} in August 2007.

The density of *Daphnia* spp. (Figure 3.16b) did not exceed 6 ind. L^{-1} throughout the monitoring period, and comprised *D. hyalina* entirely. Peak abundances of *Daphnia* spp. were observed during June during 2005 and 2006, and in April 2007. *Daphnia* spp. were observed at significant densities during 2005 and 2006 for longer periods than 2007, when the density of *Daphnia* spp. only exceeded 1 ind L^{-1} for April. *Ceriodaphnia* spp. (Figure 3.16c) were rare members of the zooplankton community throughout the monitoring period. Except for during May 2005, *Ceriodaphnia* spp.

were mainly observed during November and December in all years. Calanoid copepods (Figure 3.16d) were also rarely observed in Lake C2. Isolated occurrences were recorded during July and November 2005, although at very low densities. Cyclopoid copepods (Figure 3.16e) were the most abundant member of the zooplankton community during 2005 and 2006, exceeding 40 ind. L^{-1} during August in both years. During summer 2007, cyclopoid copepods were comparatively rare, although still contributed significantly to the total zooplankton abundance. The *Bosmina* genus (Figure 3.16f) was represented only by *B. longirostris*. The abundance of *B. longirostris* increased gradually throughout the summers of 2005 and 2006, reaching maximum abundances of 5 ind. L^{-1} and 12 ind. L^{-1} in 2005 and 2006 respectively. *B. longirostris* was observed throughout 2007, although at lower densities than during 2005 or 2006, and contributed significantly to total zooplankton abundance. The majority of rotifers (Figure 3.16g) observed were *Asplancha* spp. *Keratella* spp. were observed in April 2006 and 2007, and late summer 2007. The density of rotifers did not exceed 2 ind. L^{-1} throughout 2005 and 2006. During 2007 the density of rotifers reached 5 ind. L^{-1} in August.

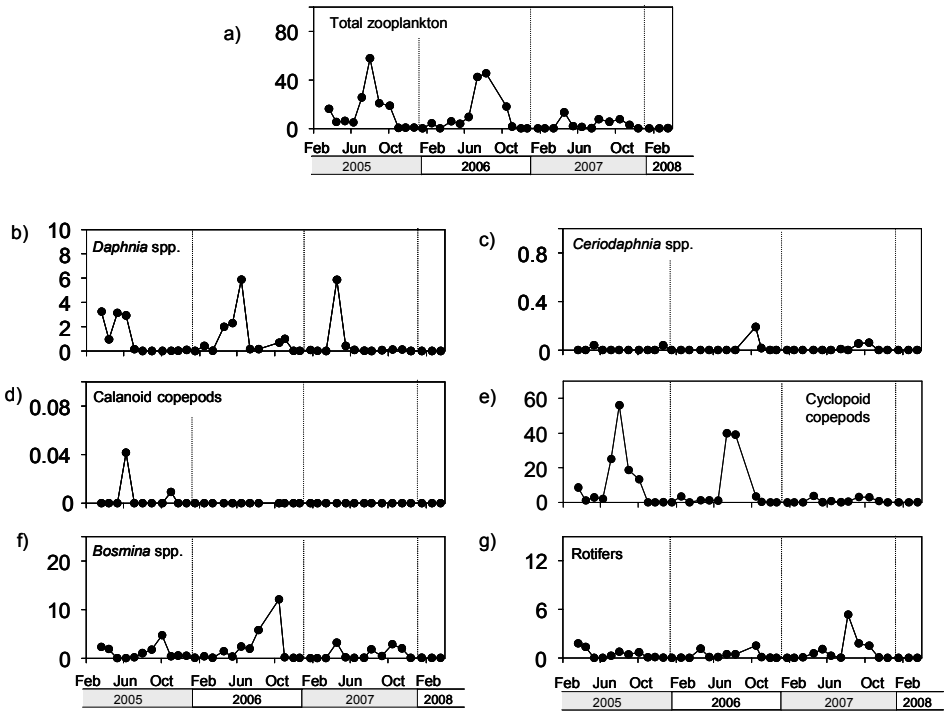


Figure 3.16 Timeseries of zooplankton abundance: a) total abundance; b-g) selected groups, March 2005-March 2008 in Lake C2 (Tween Pond). All data are presented as individuals L^{-1}.

3.4 Lake C3 (Main Pond)

Figure 3.17 A view across Lake C3 (Main Pond), approximately north, September 2008. Lake C3 is the final lake in the chain connected to the River Erewash. The aggregate processing plant is beyond the left-hand side of the photograph, and the outflow to the River Trent (MPO), not visible in the photograph, is towards the northern end of the lake.

3.4.1 Water chemistry

TP concentrations in Lake C3 (Figure 3.17) ranged between 196 µg L^{-1} and 1261 µg L^{-1} (Figure 3.18a). Mean TP was 506 (±35) µg L^{-1} during the monitoring period. TP concentrations rose from the start of the monitoring period (240 µg L^{-1}) to July 2007 (987 µg L^{-1}), before declining gradually until April 2004. TP then rose sharply from 407 µg L^{-1} in June to 1261 µg L^{-1} in August 2006, declining throughout the winter and spring to 196 µg L^{-1} in March 2007. Concentrations rose slightly throughout the summer of 2007, although maximum concentrations during 2007 were lower than in 2005 and 2006 and occurred later in the year (556 µg L^{-1}, October 2007). The mean SRP concentration during the monitoring period was 326 (±24 µg L^{-1}), ranging between 82 and 724 µg L^{-1}. SRP concentrations (Figure 3.18a) during 2005 and 2006

increased rapidly from approximately 100 µg L^{-1} in April to reach summertime peaks in July 2005 and September 2006. Progressive falls in SRP concentrations occurred between autumn and the beginning of the subsequent summer. Summertime peaks of SRP decreased each year, from 724 µg L^{-1} in 2005 to 379 µg L^{-1} in 2007.

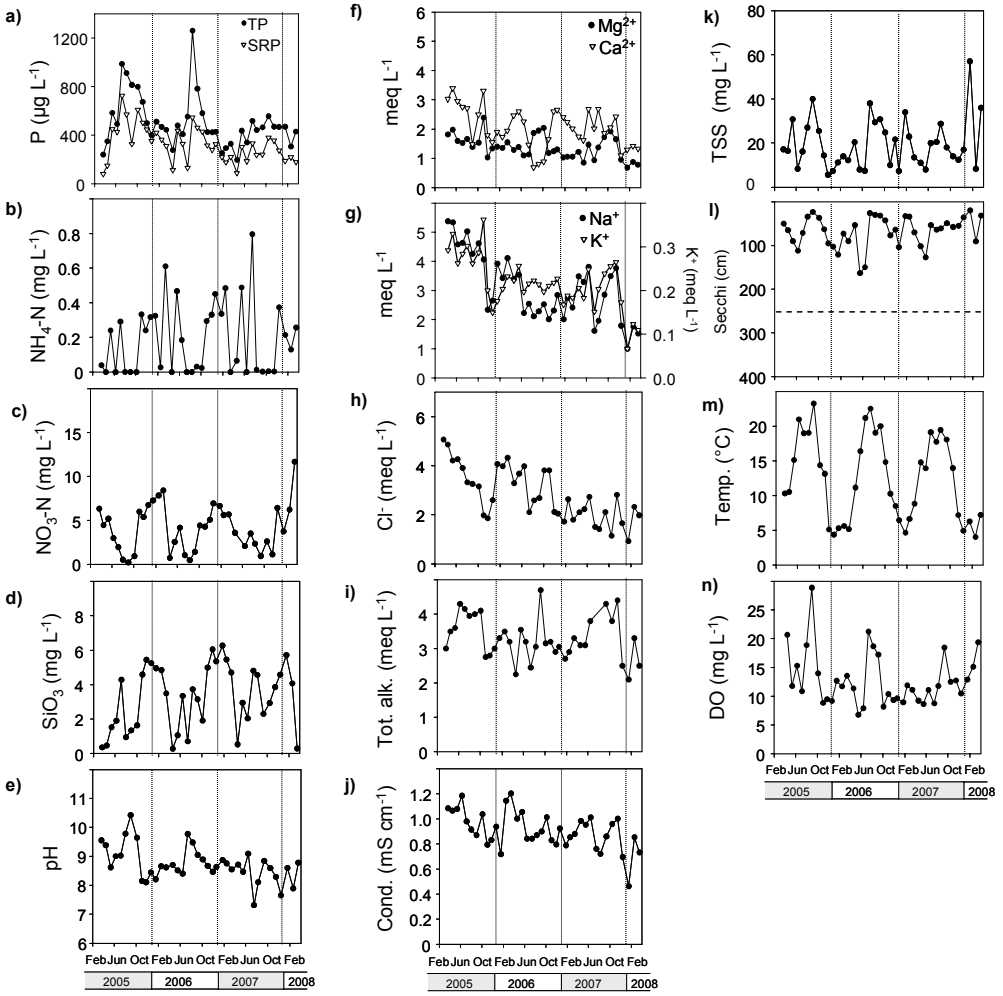


Figure 3.18 Water chemistry and physico-chemical properties of Lake C3 (Main Pond), March 2005 to March 2008: a) TP and SRP (µg L^{-1}); b) NH_4-N (µg L^{-1}); c) NO_3-N (mg L^{-1}); d) SiO_3 (mg L^{-1}); e) pH; f) Mg^{2+} and Ca^{2+} (meq L^{-1}); g) Na^+ and K^+ (meq L^{-1}); h) Cl^- (meq L^{-1}); i) total alkalinity (meq L^{-1}); j) specific conductivity (mS cm^{-1}); k) TSS (mg L^{-1}); l) Secchi disk depth (cm, dotted line represents approximate lake depth); m) surface temperature (°C); n) DO (mg L^{-1}).

NH_4-N concentrations (Figure 3.18b) during the monitoring period ranged between 0 and 0.80 mg L^{-1} (mean, 0.19 ±0.03 mg L^{-1}). NH_4-N concentrations were highly seasonal, characterised by two sharp increases and decreases in early summer,

concentrations below detectable limits during late summer, and then significant increases and sustained higher concentrations throughout the winter. In 2007, summer peaks were around 0.5 mg L^{-1} greater than in 2005 and 0.2 mg L^{-1} than in 2006, and occurred later in the year, although the concentrations of NH_4-N during winter 2007 were lower than in previous years. The mean NO_3-N concentration during the monitoring period was 4.1 (±0.4) mg L^{-1} and ranged between 0.2 and 11.7 mg L^{-1} (Figure 3.18c). Concentrations of NO_3-N were highest between October and March, reaching 8.4 mg L^{-1} in March 2006 and 6.9 mg L^{-1} in December 2006. NO_3-N concentrations in 2005 and 2007 decreased gradually throughout summer, although in 2006 a rapid decline in NO_3-N occurred between March and April. From November 2007, NO_3-N concentrations increased steeply until March 2008.

The mean SiO_3 concentration during the monitoring period was 3.3 (±0.3) mg L^{-1}. SiO_3 concentrations were highly seasonal (Figure 3.18d). Concentrations typically rose through spring and early summer to between 4 and 5 mg L^{-1} in June to July, before decreasing briefly throughout late summer and then increasing rapidly again in winter. Wintertime concentrations were typically > 5 mg L^{-1}. During 2007, SiO_3 concentrations were not as high in winter as in 2005 and 2006, and a rapid decrease was observed from January 2008 to March 2008 (from 5.7 to 0.2 mg L^{-1}).

pH (Figure 3.18e) during the monitoring period ranged between 7.31 and 10.42. Mean pH was 8.73 (±0.1) and generally decreased during the monitoring period (r_s = -0.403, p = 0.011). pH was highest in August 2005 (10.42) and June 2006 (9.77). During summer 2007, pH fell to 7.31 after reaching a maximum of 9.09 in June 2007. pH was lowest during autumn and winter, ranging between 8.10 and 8.70 during winter 2005 and 8.46 and 8.87 during winter 2006. Throughout the majority of the monitoring period, pH was homogenous throughout the water column (Figure 3.19a). Some of the strongest gradients in pH were observed during late summer 2005, when pH at the water surface was 10.4 and decreased with increasing depth to reach 9.14 at 2.5 m.

Mg^{2+} concentrations (Figure 3.18f) ranged between 0.68 and 2.39 meq L^{-1} (mean, 1.39 ±0.64 meq L^{-1}) during the monitoring period and showed a declining trend (r_s = -0.470, $p \leq 0.003$). The highest Mg^{2+} concentrations occurred during November 2005 (2.39 meq L^{-1}), October 2006 (2.04 meq L^{-1}) and October 2007 (1.92 meq L^{-1}). The

mean Ca^{2+} concentration during the monitoring period was 2.05 (±0.11) meq L^{-1}, and ranged between 0.70 and 3.39 meq L^{-1} (Figure 3.18f). Ca^{2+} concentrations were highest during April and November 2005 (3.39 and 3.30 meq L^{-1} respectively). During 2006 and 2007 Ca^{2+} concentrations did not exceed 2.7 meq L^{-1}. A substantial decrease was observed during late summer and autumn 2006. Ca^{2+} concentrations generally declined during the monitoring period (r_s -0.413, p = 0.01).

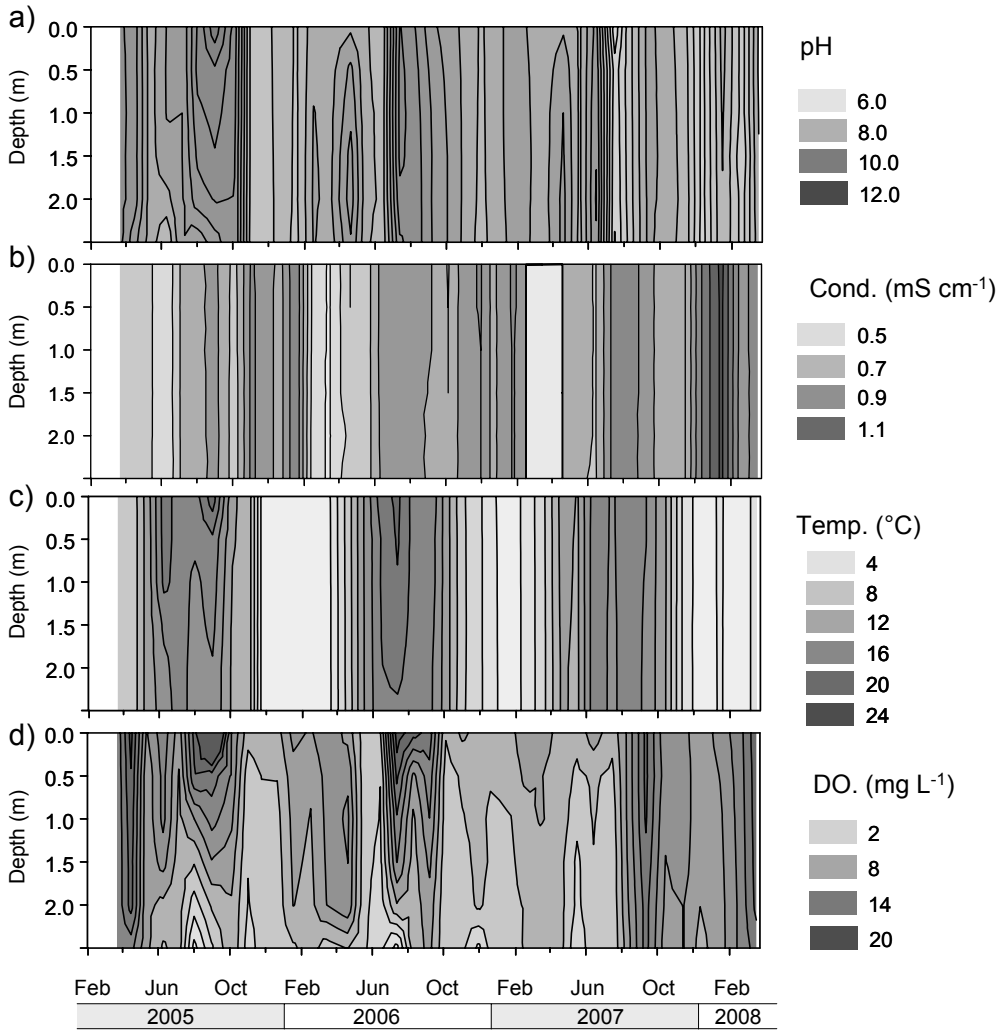


Figure 3.19 Profiles of a) pH; b) specific conductivity (mS cm^{-1}); c) temperature (°C); d) dissolved oxygen (mg L^{-1}) in Lake C3 (Main Pond) March 2005 to March 2008.

A trend towards decreasing Na^+ concentrations was observed throughout the monitoring period (Figure 3.18g, r_s = -0.713, $p \leq 0.001$). The mean Na^+ concentration was 3.10 (±0.18) meq L^{-1}, and ranged between 1.00 and 5.38 meq L^{-1}. The highest concentrations of Na^+ occurred during spring and early summer each year, reaching

5.37 meq L^{-1} in March 2005, 4.11 meq L^{-1} in March 2006 and 3.81 meq L^{-1} in June 2007. Na^+ concentrations rose steeply between July and November (from 1.62 to 3.76 meq L^{-1}) and then fell sharply again from December 2007 to March 2008. K^+ concentrations (Figure 3.18g) showed a trend of decreasing concentrations throughout the monitoring period (r_s = -0.630, $p \leq 0.001$). The mean K^+ concentration was 0.22 (±0.01) meq L^{-1}, and ranged between 0.07 and 0.36 meq L^{-1}. The highest concentrations of K^+ occurred during November 2005 and November 2007 (0.36 and 0.26 meq L^{-1}) respectively. During 2006, K^+ concentrations were highest in May.

Cl^- concentrations (Figure 3.18h) decreased throughout the monitoring period (r_s = -0.705, $p \leq 0.001$). The mean Cl^- concentration was 2.76 (±0.17) meq L^{-1} and ranged between 0.92 and 4.86 meq L^{-1}. Concentrations declined from March (5.06 meq L^{-1}) to November 2005 (1.85 meq L^{-1}) and then rose sharply to 4.07 meq L^{-1} by January 2006. Cl^- concentrations then became more variable and oscillated throughout the rest of the monitoring period. Total alkalinity (Figure 3.18i) ranged between 2.10 and 4.70 meq L^{-1} (mean, 3.31 ± 0.10 meq L^{-1}). Total alkalinity was highest between July and October, although in 2006, the summertime maximum was considerably shorter in duration than 2005 and 2007.

Mean specific conductivity during the monitoring period was 0.91 (±0.02) mS cm^{-1}, ranging between 0.47 and 1.20 mS cm^{-1} (Figure 3.18j). Specific conductivity showed a decreasing trend over the monitoring period (r_s = -0.525, p = 0.001). Seasonal trends in specific conductivity were not clearly evident. High specific conductivity was recorded during June 2005 (1.18 mS cm^{-1}) and March 2006 (1.20 mS cm^{-1}). During 2007, the highest specific conductivity was 1.01 mS cm^{-1} in June. Specific conductivity did not show any substantial changes with increasing water depth (Figure 3.19b).

TSS concentrations (Figure 3.18k) ranged between 5.6 and 57.0 mg L^{-1} (mean, 19.6 ±1.8 mg L^{-1}). TSS concentrations were higher in summer during 2005 and 2006 (40 and 38 mg L^{-1} respectively), and in each year rose and fell briefly during early summer. TSS concentrations increased sharply to 34 mg L^{-1} in January 2007 then fell gradually to 8 mg L^{-1} in May 2007. Summertime concentrations of TSS in 2007 were less than in both 2005 and 2006, reaching 28.8 mg L^{-1} in August, and rose sharply to

57.0 mg L^{-1} in January 2008. TSS concentrations rose briefly in May 2005, and after declining in June, rose sharply to 40.0 mg L^{-1} in August 2005.

Secchi disk depths (Figure 3.18l) ranged between 20 and 163 cm in Lake C3. The mean Secchi depth for the monitoring period was 68 (±6) cm. The greatest Secchi disk depths tended to occur during January and May each year. During 2006, early summer increases in Secchi depth were more rapid than in 2005, and a greater depth was recorded (112 cm, June 2005; 163 cm, May 2006). Secchi depth tended to decreased during later summer, reaching 23 cm and 26 cm in 2005 and 2006 respectively. In 2007, low Secchi disk depths were observed during January and February (33-34 cm), and although early summer increases in Secchi depth were observed, the minimum late summer Secchi depth was not as low as in previous years, reaching 49 cm in September 2007.

Surface water temperature (Figure 3.18m) was highly seasonal. Maximum temperatures occurred during August or September each year. Minimum temperatures were recorded between December and January each year. The mean temperature throughout the monitoring period was 12.6 (±1.0)°C. Water temperature was homogenous throughout the water column for the majority of the monitoring period (Figure 3.19c). Temperature gradients from the surface to the bottom of the lake only occurred during the summers of 2005 and 2006. In August 2005, the water temperature decreased from 23.3°C on the surface to 17.7°C at 2.5 m depth, and in July 2006 surface water temperature was 2.8°C warmer than at 2.5 m depth.

DO concentrations in Lake C3 (Figure 3.18n) ranged between 6.8 and 30.0 mg L^{-1} (mean, 13.5 ±0.9 mg L^{-1}) during the monitoring period. Peaks in DO concentration were recorded during August 2005 and July 2006 and September 2007. During winter months, DO typically ranged between 7 and 14 mg L^{-1}. DO concentrations began to increase from December 2007 to March 2008. DO concentrations were frequently greater nearer the water surface than further down the water column (Figure 3.19d). This was particularly evident during summer 2005 and 2006. In both years, DO concentrations were around 20 mg L^{-1} greater at the surface than 2.5 m deep.

3.4.2 Phytoplankton

Total biomass

Chlorophyll-*a* concentrations ranged between 3 and 286 µg L^{-1} (mean 68 ±13 µg L^{-1}) during the monitoring period. Chlorophyll-*a* concentrations in 2005 and 2006 were strongly seasonal. In both years, chlorophyll-*a* concentrations decreased from 91-123 µg L^{-1} during April to < 8 µg L^{-1} in May and June, followed by rapid rises in chlorophyll-*a* concentrations to reach 281 and 286 µg L^{-1} in August 2005 and 2006 respectively. Chlorophyll-*a* concentrations decreased rapidly to < 8 µg L^{-1} in December. In 2007, an early summer increase in chlorophyll-*a* concentrations was not observed, and the highest concentration of chlorophyll-*a* was less than half that of the previous two years.

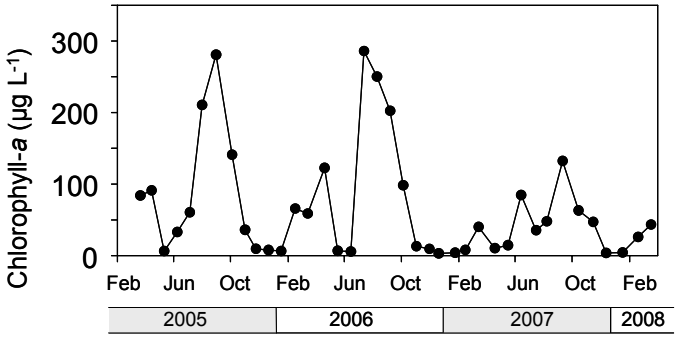


Figure 3.20 Chlorophyll-*a* concentrations in Lake C3 (Main Pond), March 2005 to March 2008.

Community composition

Chlorophytes and diatoms were the most abundant phytoplankton groups in Lake C3 during summer months (Figure 3.21), particularly during 2005 and 2006. During summer 2007, the phytoplankton community was substantially smaller than in previous summers and was not clearly dominated by any phytoplankton group. Cryptophytes and cyanobacteria were also abundant during the summer months. Cyanobacteria were rarely observed outside of summer months. During spring, chlorophytes were more abundant in 2005 and diatoms were dominant in 2006 and

2007. Chrysophytes and dinophytes were observed in the phytoplankton during the monitoring period, although overall were rare compared to diatoms and chlorophytes.

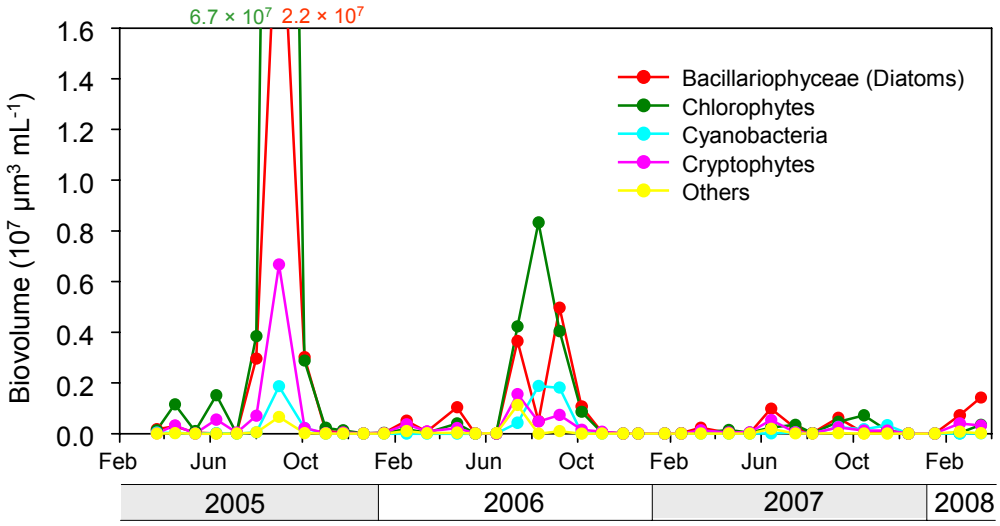


Figure 3.21 Summary of total phytoplankton group biovolumes in Lake C3 (Main Pond), March 2005 to March 2008. 'Bacillariophyceae' refers to all diatoms, and 'others'comprises dinophytes and chrysophytes.

Diatoms (Figure 3.22a-e) were present throughout the monitoring period in Lake C3. Centric species (Figure 3.22a) were highly abundant during late summer before falling sharply in abundance during October and November. In 2005 and 2006, increases in the biovolume of centric species were observed in spring. The maximum abundance of centric species in 2006 was substantially less than in 2005, reaching 3.4 × 10^6 µm^3 mL^{-1} in July 2006. Centric diatoms were less abundant in 2007 than either 2005 or 2006. From January 2008 to March 2008, the biovolume of centric diatoms rose several orders of magnitude. Pennate diatoms (principally *Navicula* spp., Figure 3.22b) were considerably less abundant throughout the monitoring period, and tended to be present later in the year than centric species. During 2005, pennate species were only present in significant biovolumes during October 2005 and in September 2006 reached 2.2 × 10^6 µm^3 mL^{-1}. Pennate diatoms were again present in the phytoplankton community during September 2007, although in low biovolumes compared to previous years. *Asterionella* sp. (Figure 3.22c) was largely absent for most of the monitoring period. *Aulacoseira* spp. (Figure 3.22d) were also often observed in the phytoplankton, although did not reach significant biovolumes apart from briefly during the summer of 2005 (1.4 × 10^6 µm^3 mL^{-1}). *Synedra* spp. (Figure

3.22e) were often observed although were not present in large biovolumes regularly, except during April 2006. *Tabellaria* sp. (data not shown) was recorded during October 2006, September 2007 and in March 2008, although in relatively low biovolumes compared to other diatom species.

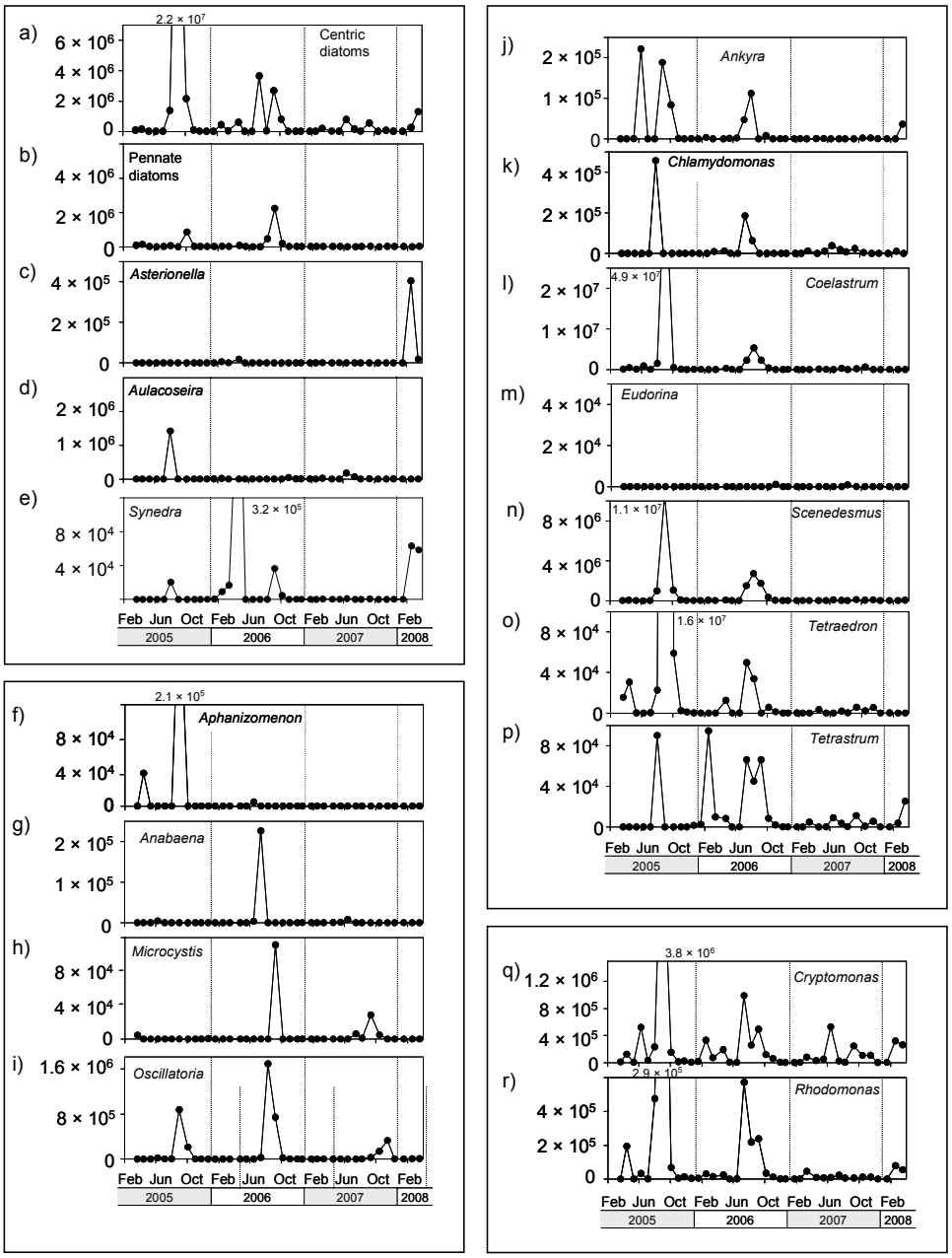


Figure 3.22 Timeseries of biovolumes (µm^3 mL^{-1}) of selected phytoplankton species in Lake C3 (Main Pond), March 2005-March 2008: a-e) Bacillariophyceae (diatoms); f-i) cyanobacteria; j-p) Chlorophyceae; q-*r*) Cryptophyceae. Note the differences in *y*-axis scales.

Cyanobacteria (Figure 3.22f-i) were more abundant during the summer of 2006 than other years in Lake C3. During 2005, only *Aphanizomenon* sp. (Figure 3.22f) and *Oscillatoria* sp. (Figure 3.22i) were present in significant biovolumes. *Aphanizomenon* sp. was present in the phytoplankton during spring 2005, however during 2006 and 2007 it was very scarce. *Microcystis* sp. (Figure 3.22h) was abundant during September 2006. *Microcystis* sp. was also present in the phytoplankton during autumn 2007 with *Oscillatoria* sp., although in much smaller quantities than during 2006. *Merismopedia* sp. was present in comparatively high biovolumes during August 2005 (7.9×10^5 μm^3 mL^{-1}, data not shown).

Chlorophyte species (Figure 3.22j-p) were abundant through both 2005 and 2006 in Lake C3. The genus *Ankyra* (Figure 3.22j) was represented by *A. judyaii* alone, reaching a biovolume of 2.2×10^5 μm^3 mL^{-1} in June 2005. *A. judyaii* was frequently present during winter 2005 and spring 2006 and spring 2007. *Chlamydomonas* spp. (Figure 3.22k) showed similar patterns of abundance to *Ankyra* spp. *Chlamydomonas* spp. were abundant briefly during August 2005 (4.6×10^5 μm^3 mL^{-1}). During summer 2006, *Chlamydomonas* spp. were observed for a longer period (March to August 2006) although in smaller biovolumes compared to 2005. During 2007, *Chlamydomonas* spp. were observed throughout spring and summer, at biovolumes that did not exceed 2.4×10^4 μm^3 mL^{-1}. *Coelastrum* sp. (Figure 3.22l) was consistently observed in the phytoplankton community from March to November 2005. *Coelastrum* sp. was more abundant during 2005 than 2006 and was a significant contributor to the total biomass of chlorophyte species during 2007. *Eudorina* sp. (Figure 3.22m) was rare throughout the monitoring period. *Scenedesmus* spp. (Figure 3.22n) were represented by *S. communis*, *S. falcatus* and *S. opoliensis* and were observed throughout the monitoring period. *Scenedesmus* spp. were particularly abundant during 2005 largely driven by high biovolumes of *S. falcatus* and *S. communis* in August 2005.. *S. falcatus* and *S. communis* were the most abundant species during summer 2006. *Scenedesmus* spp. were comparatively rare during 2007, although were the most abundant genera during summer 2007. *Tetraedron* spp. (Figure 3.22o) were frequently observed in the phytoplankton community. *T. triangulare*, *T. regulare* and *T. caudatum* were the most frequently observed members of the genus. The maximum biovolume of *Tetraedron* spp. during 2007 was substantially less than in previous years. *Tetrastrum* sp. (Figure 3.22p) was also frequently observed in the

phytoplankton community of Lake C3 during late summer. Other chlorophytes that formed a significant biovolume of the phytoplankton included *Closterium acutum*, which was present during 2005, largely absent throughout 2006, and present during 2007. *Dictyosphaerium* sp. was observed in the summer of 2005 only, and reached high biovolumes (4.2×10^6 μm^3 mL^{-1}) in August. *Monoraphidium* sp. was observed during early summer and autumn each year, although it was rarely a significant member of the chlorophyte community. *P. duplex* and *P. simplex* were both observed during the monitoring period throughout the summer months in all years, although the highest biovolumes occurred during 2006 (7.5×10^4 μm mL^{-1} in June).

Cryptomonas spp. (Figure 3.22q) were observed throughout the monitoring period. The highest biovolumes of *Cryptomonas* spp. occurred during August 2005 and July 2006 and 2007, although the highest biovolume recorded fell each year. *Rhodomonas* spp. (Figure 3.22r) were also observed throughout the majority of the monitoring period, and although present in smaller biovolumes than *Cryptomonas* spp., were most abundant during 2005. In 2006, the greatest biovolume of *Rhodomonas* spp. were significantly less during 2005, reaching a maximum of 5.7×10^5 μm^3 mL^{-1} in July. During 2007, *Rhodomonas* spp. were rare with the maximum abundance of the genus reaching an order of magnitude less than in 2006 (4.6×10^4 μm mL^{-1} in February). An increase in the biovolume of *Rhodomonas* spp. occurred from January 2008 to March 2008.

Euglenophytes (data not shown) were generally rare throughout the monitoring period. However, *Trachelomonas* sp. was observed on several occasions throughout the monitoring period, particularly during 2005 and 2006. Dinophytes were represented by *Gymnodinium* spp. and *Peridinium* spp. in Lake C3, and were mainly observed during summer. During 2007, dinophytes were not as abundant as in 2006. *Mallomonas* sp. was observed in relatively low biovolumes during 2006 and 2007.

3.4.3 Macrophytes

Submerged macrophytes were very rare in Lake C3 (Figure 3.23). *Zanichella palustris* was found towards the north-east of the lake and filamentous algae were found at seven sampling sites. The floating-leaved species *Nuphar lutea* and *Nymphea alba*

were found along the north-western shore, and *Polygonum amphibium* was observed twice, along the north and south shores.

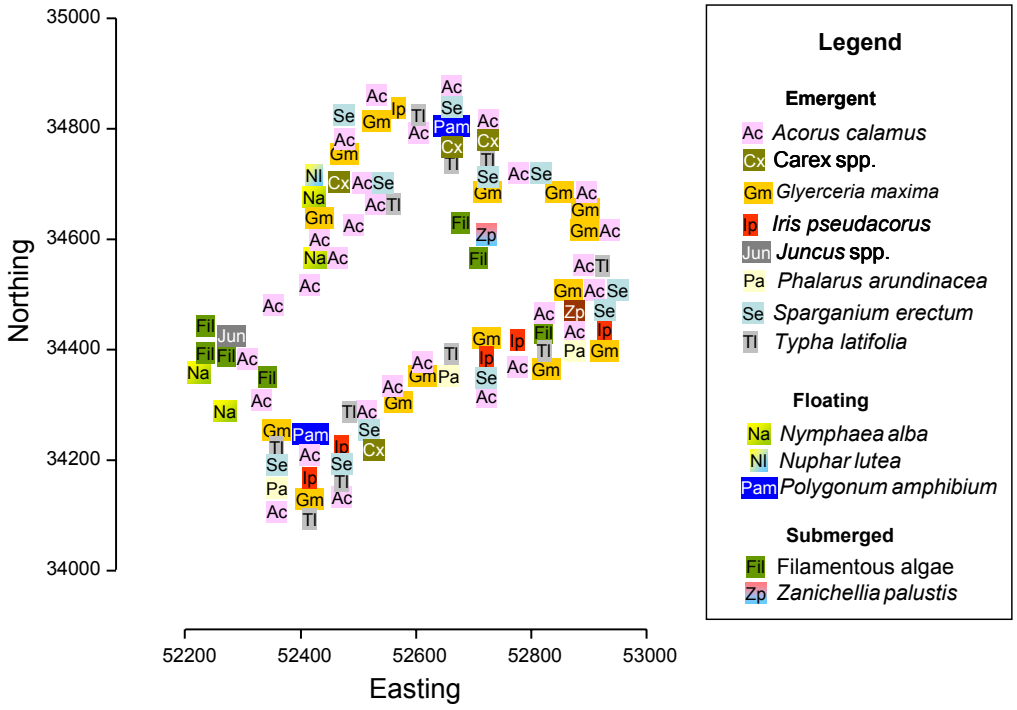


Figure 3.23 Macrophytes observed in Lake C3 (Main Pond) during summer 2005.

The emergent macrophyte community of Lake C3 comprised eight species. *Glyceria maxima*, *Acorus calamus* and *Sparganium erectum* were the most frequently observed emergent species, occurring along all the shores of the lake. *Typha latifolia* was also frequently recorded. *Carex* sp. was found along the northern-most shore, and *Juncus* sp. along the western edge of Lake C3. *Iris pseudacorus* was found along the southern edge of the lake and at one sampling site on the northern shore. *Phalaris arundinacea* was also found on the southern edge of the lake.

3.4.4 Zooplankton

Total zooplankton abundance (Figure 3.24a) in Lake C3 decreased during early summer 2005 from 52 ind. L^{-1} to < 2 ind. L^{-1} by June 2005. Total zooplankton abundance increased throughout late summer and decreased during winter. Total zooplankton abundance remained low during winter 2005 until a brief increase in April 2006, followed by a rapid increase in abundance from 8 ind. L^{-1} in June 2006 to 90 ind. L^{-1} in September 2006. Total zooplankton abundance then declined sharply to

<0.1 ind. L^{-1} and remained low until March 2007, before decreasing slightly between April and May. The highest abundance of zooplankton in 2007 occurred in June, after which a sharp decrease in total abundance occurred. An increase in total zooplankton abundance was observed between July and October. Fewer than 4 ind. L^{-1} were present from November 2007 until March 2008.

In Lake C3, the genus *Daphnia* (Figure 3.24b) was almost exclusively dominated by *D. hyalina*, except for an isolated occurrence of *D. cucullata* in March 2005. The abundance of *Daphnia* spp. was greatest during early summer in 2005 and 2006. A sharp increase in *Daphnia* spp. abundance occurred between May and June 2006. *Daphnia* spp. were less abundant during 2007 than in previous years. *Ceriodaphnia* spp. (Figure 3.24c) were absent throughout most of the monitoring period.

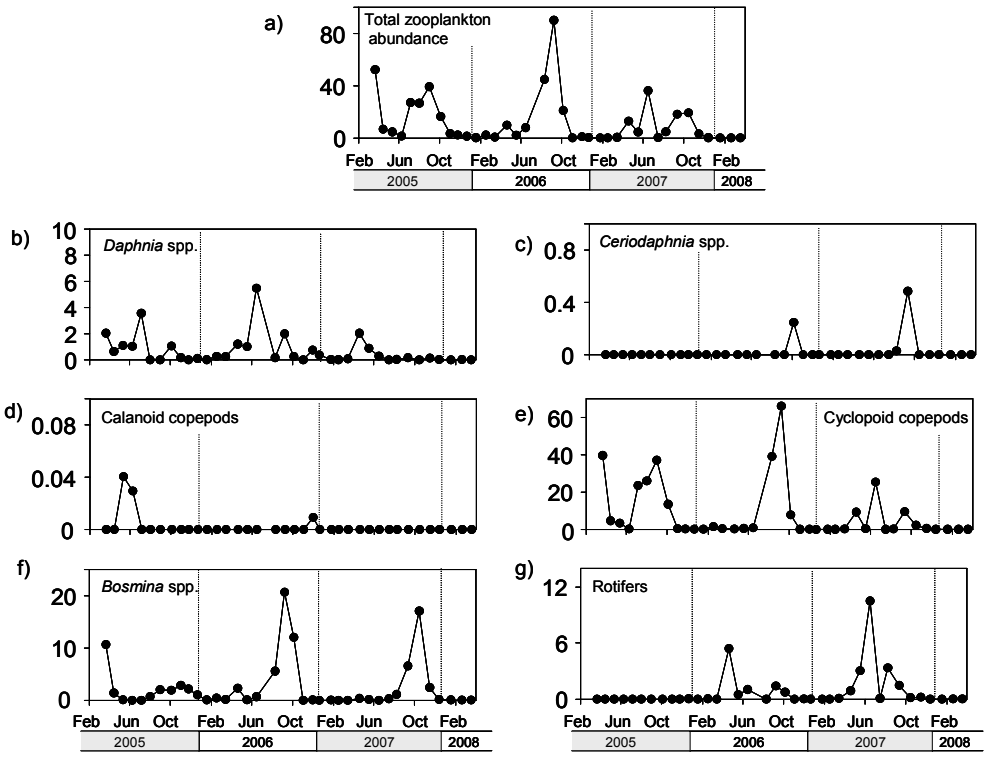


Figure 3.24 Timeseries of zooplankton abundance: a) total abundance; b-g) selected groups, March 2005-March 2008 in Lake C3 (Main Pond). All data are presented as individuals L^{-1}.

Calanoid copepods (Figure 3.24d) were rarely observed in Lake C3. Cyclopoid copepods (Figure 3.24e) were the most abundant zooplankton group. Cyclopoid copepods were most abundant during spring and summer 2005 and summer 2006. The abundance of cyclopoid copepods rose to 66 ind. L^{-1} during autumn 2006. The abundance of calanoid copepods during 2007 was highest in June. Low densities were observed during July and August. The genus *Bosmina* (Figure 3.24f) was represented by *B. longirostris* alone. *B. longirostris* was substantially more abundant (exceeding 15 ind. L^{-1}) during late summer 2006 and 2007 than 2005.

Rotifers found in zooplankton samples (Figure 3.24g) were largely *Asplancha* sp., although *Keratella* sp. was also observed. Rotifers were generally not found in any samples during 2005. 5 ind. L^{-1} were recorded in April 2006. Rotifers remained rare until June 2007, when a sharp increase in abundance was observed, reaching 11 ind. L^{-1}. After June 2007, rotifer abundance fell and remained <1 ind. L^{-1} from September until the end of the monitoring period. *Eurycercus lamellatus* was recorded in April 2006 and March 2008, and *Sida* spp. in April 2007 (data not shown).

3.5 Lake I1 (Church Pond)

Figure 3.25 A north-easterly view across Lake I1 (Church Pond), September 2008. Lake I1 is isolated from the River Erewash. To the right of the photograph is a platform to encourage nesting wildfowl, particularly common terns (*Sterna hirunda*).

3.5.1 Water chemistry

In Lake I1 (Figure 3.25) TP concentrations ranged between 69 and 725 µg L^{-1}, and SRP concentrations ranged between 14 and 319 µg L^{-1} (Figure 3.26a and b). Mean concentrations of TP and SRP were 192 (± 21) and 103 (±12) µg L^{-1} respectively. Both TP and SRP were present in higher concentrations during late summer than in other seasons and at their lowest during spring. Highest concentrations of TP (725 µg L^{-1}) occurred in August 2006, although the highest SRP concentrations (319 µg L^{-1}) were recorded in August 2005. Both SRP and TP concentrations were lower during the summer of 2007 than the previous two years.

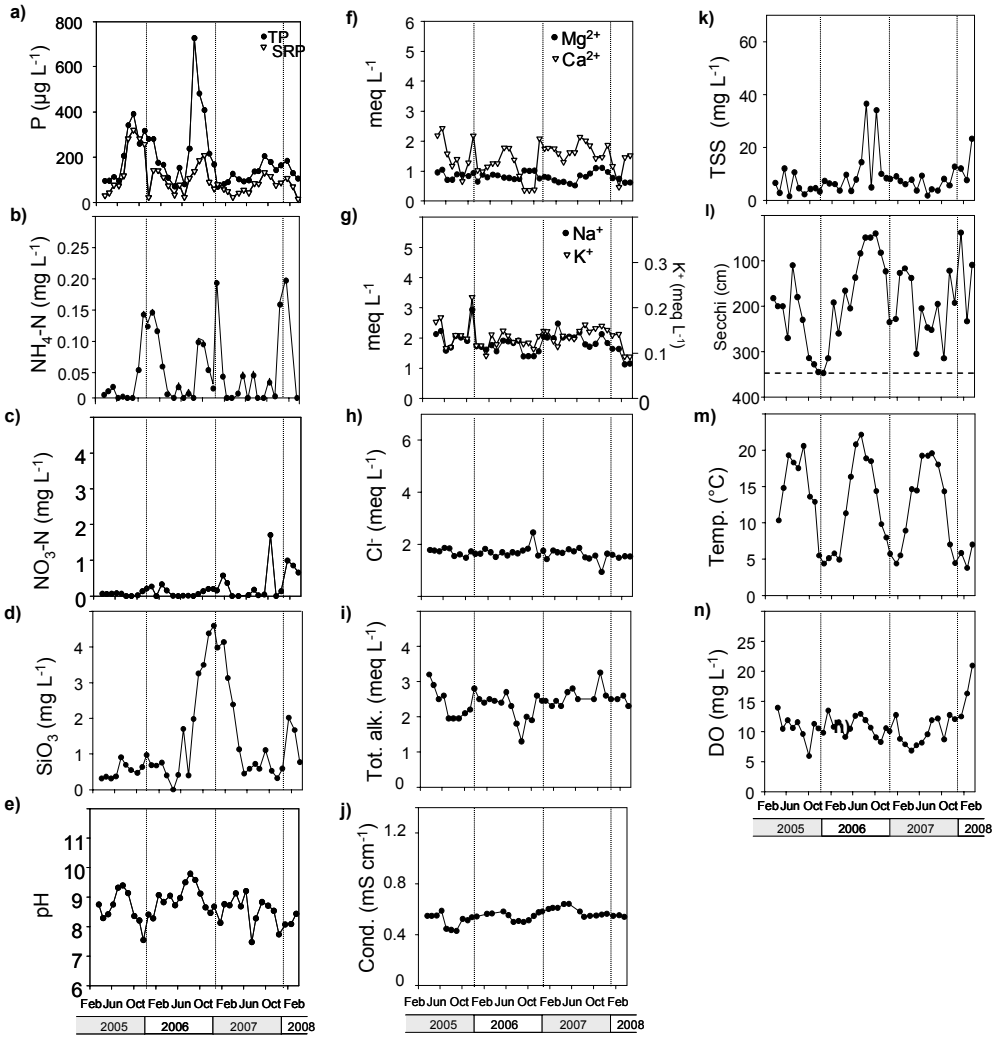


Figure 3.26 Water chemistry and physico-chemical properties of Lake I1 (Church Pond), March 2005 to March 2008: a) TP and SRP (µg L^{-1}); b) NH_4-N (mg L^{-1}); c) NO_3-N (mg L^{-1}); d) SiO_3 (mg L^{-1}); e) pH; f) Mg^{2+} and Ca^{2+} (meq L^{-1}); g) Na^+ and K^+ (meq L^{-1}); h) Cl^- (meq L^{-1}); i) total alkalinity (meq L^{-1}); j) specific conductivity (mS cm^{-1}); k) TSS (mg L^{-1}); l) Secchi disk depth (cm, dotted line represents approximate lake depth); m) surface temperature (°C); n) DO (mg L^{-1}).

The mean NH_4-N concentration was 0.05 (±0.01) mg L^{-1} during the monitoring period (Figure 3.26b) and concentrations were frequently below detectable levels. NH_4-N concentrations were highly seasonal, highest during late winter (in excess of 0.02 mg L^{-1}) and less than 0.05 mg L^{-1} during summer. During summer 2007 concentrations were slightly higher than previous summers. NO_3-N concentrations ranged between 0 and 1.7 mg L^{-1}, with a mean of 0.2 ±0.06 mg L^{-1} over the monitoring period (Figure 3.26c). Concentrations of NO_3-N were below detectable limits during

early summer in 2005 and 2006 (April and May 2006, March and April 2007). During 2005 and 2006, NO_3-N concentrations increased from below detectable concentrations in late summer, to winter maxima of 0.3 and 0.6 mg L^{-1} respectively. In 2007, NO_3-N concentrations rose abruptly in October to 1.7 mg L^{-1}, and remained higher in January to March 2008 than previous years.

The mean SiO_3 concentration in Lake I1 was 1.4 ±0.2 mg L^{-1} and ranged between 0.3 and 4.6 mg L^{-1} (Figure 3.26d). Strong inter-annual variability was observed in SiO_3 concentrations. Wintertime SiO_3 concentrations during 2005 and 2007 were 1.0 and 1.1 mg L^{-1} respectively, although higher concentrations were recorded during the winter of 2006, rising from 0.4 mg L^{-1} in June 2006 and reaching a maximum of 4.6 mg L^{-1} in January 2007. After January 2006, SiO_3 concentrations decreased abruptly to 0.5 mg L^{-1} in May 2007, and remained less than 1 mg L^{-1} until January 2008 when concentrations reached 2 mg L^{-1} before declining to 0.8 mg L^{-1} by March 2008.

pH ranged between 7.45 and 9.79 (mean 8.67 ±0.08, Figure 3.26e). During 2005 and 2006, pH tended to be higher during late summer and lowest during winter. In 2007, pH was not clearly higher during the summer, although decreased during the winter similarly to previous years. No substantial changes of pH with depth were observed (Figure 3.27a).

Mg^{2+} concentrations ranged between 0.51 and 1.11 meq L^{-1} during the monitoring period with a mean of 0.81 ±0.02 meq L^{-1} (Figure 3.26f). Mg^{2+} concentrations remained relatively constant throughout 2005, although during 2006 and 2007 concentrations tended to be higher during late summer. The mean Ca^{2+} concentration was 1.42 ±0.09 meq L^{-1}, ranging between 0.35 and 2.44 meq L^{-1}. Seasonality in concentrations was not evident (Figure 3.26f), although during winter 2006 and January 2008 Ca^{2+} concentrations were markedly low. Na^{+} concentrations ranged between 1.13 and 2.94 meg L^{-1} with a mean of 1.83 ±0.06 meq L^{-1} (Figure 3.26g). K^{+} concentrations ranged between 0.09 and 0.22 meq L^{-1}, with a mean of 0.14 ± 0.004 meq L^{-1} (Figure 3.26g). Na^{+} and K^{+} concentrations followed similar trends throughout the monitoring period. Noticeable peaks occurred during November 2005. Concentrations were relatively constant during 2006 and 2007 and fell from September 2007 towards the end of the monitoring period. Cl^{-} concentrations

remained relatively constant throughout the monitoring period (Figure 3.26h). The mean Cl^- concentration was 1.65 ±0.10 meq L^{-1}, and ranged between 0.70 and 2.60 meq L^{-1}. During 2005 and 2006, total alkalinity was lower in summer than other seasons (Figure 3.26i). Mean total alkalinity was 2.40 ±0.06 meq L^{-1}, ranging between 1.30 and 3.25 meq L^{-1}.

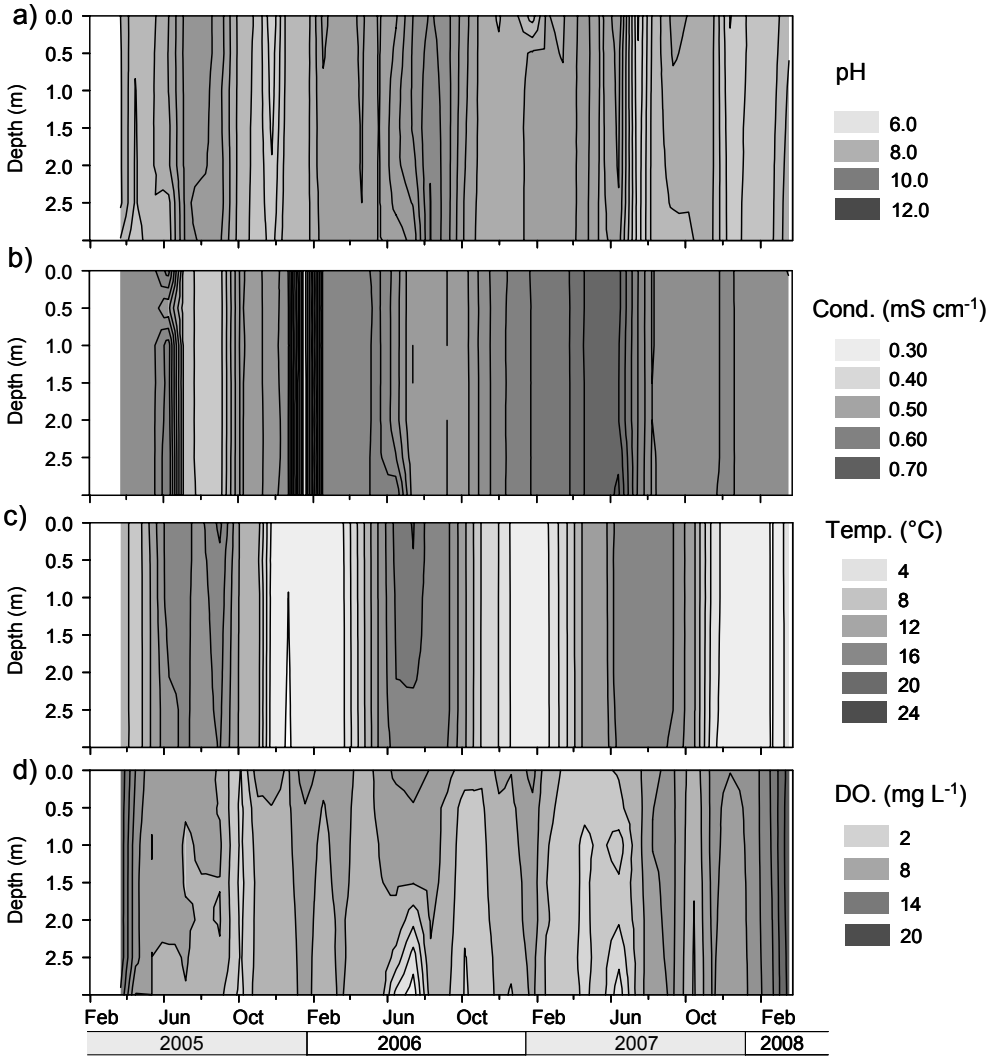


Figure 3.27 Profiles of a) pH; b) specific conductivity (mS cm^{-1}); c) temperature (°C); d) dissolved oxygen (mg L^{-1}) in Lake I1 (Church Pond) March 2005 to March 2008.

Conductivity ranged between 0.34 and 1.09 ms cm^{-1}, with a mean of 0.56 ±0.02 mS cm^{-1} during the monitoring period (Figure 3.26j). Conductivity increased from mid summer 2005 until June 2006, after which conductivity decreased briefly before increasing again to June 2006. Conductivity then decreased to the end of the

monitoring period. Conductivity was generally homogenous throughout the water column during the monitoring period (Figure 3.27b)

The mean TSS concentration during the monitoring period was 8.6 ±1.2 mg L^{-1} (Figure 3.26k). TSS concentrations were generally less than 20 mg L^{-1} during the monitoring period. However, during summer 2006 TSS increased abruptly to reach 36.5 mg L^{-1}. TSS concentrations then fell gradually to spring 2007, before increasing towards the end of monitoring period to 23.2 mg L^{-1} by March 2008.

The mean Secchi disk depth was 191 ±14 cm and ranged from 38 to 347 cm, when the disk was visible on the lake bed (Figure 3.26l). Secchi disk depths showed distinct seasonal trends and were generally greater during winter. The shallowest Secchi disk depths were recorded during summer months. Shallow Secchi disk depths were sustained during summer 2006. During summer 2007, deeper Secchi disk depths were observed than during 2006 and 2005. Secchi depths were much shallower during the winter of 2007 than previous winters.

Temperature measurements exhibited a strongly seasonal pattern (Figure 3.26m). The highest temperatures were recorded during summer months (> 19°C) and the lowest during late winter (< 5°C). The mean water temperature throughout the monitoring period was 12.2 ±1.0°C. Temperature showed no evidence of stratification of the water column (Figure 3.27c).

The mean DO concentration in Lake I2 was 11.1 ±0.5 mg L^{-1}, and ranged between 5.9 and 21.0 mg L^{-1} (Figure 3.26n). No distinct seasonality in DO concentrations was observed. From January to March 2008, DO concentrations increased steadily. DO concentrations were generally higher near the lake surface throughout the monitoring period (Figure 3.27d), although this difference rarely exceeded 2 mg L^{-1} between 0 and 3 m depth.

3.5.2 Phytoplankton

Total biomass

Mean chlorophyll-*a* concentration in Lake I1 was 23 ± 8 µg L^{-1} and ranged between 2 and 299 µg L^{-1} (Figure 3.28). In 2005 and 2006, the highest concentrations of chlorophyll-*a* were observed in late summer, although the maximum was substantially higher in 2006 than 2005. In 2007, there was no summertime maximum, instead concentrations remained less than 10 µg L^{-1} through the summer months. During 2007, chlorophyll-*a* concentrations were highest in November and again rose sharply from January to March 2008.

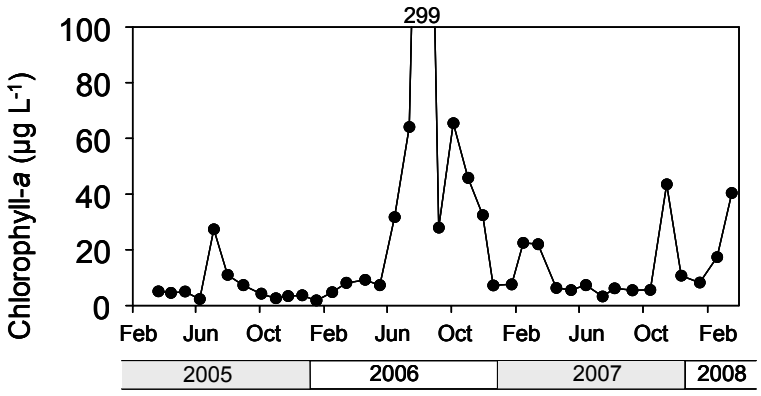


Figure 3.28 Chlorophyll-*a* concentrations in Lake I1 (Church Pond) March 2005 to March 2008.

Community composition

Cyanobacteria were the most abundant phytoplankton groups during summer 2005 and 2006 in Lake I1 (Figure 3.29). Chlorophytes were present throughout the monitoring period although they were most abundant during the summer of 2006. Cryptophytes were more abundant during 2006 and 2007, although they were present in smaller biovolumes during 2005. During 2007, maximum abundances of phytoplankton occurred later in the year than in 2005 and 2006, with euglenophytes and diatoms most abundant. Diatoms were scarce throughout 2005 and 2006, and then became an increasingly important group of phytoplankton towards the end of the monitoring period. Although dinoflagellates and chrysophytes were observed in Lake I1, they were rare compared to other phytoplankton groups during the monitoring period.

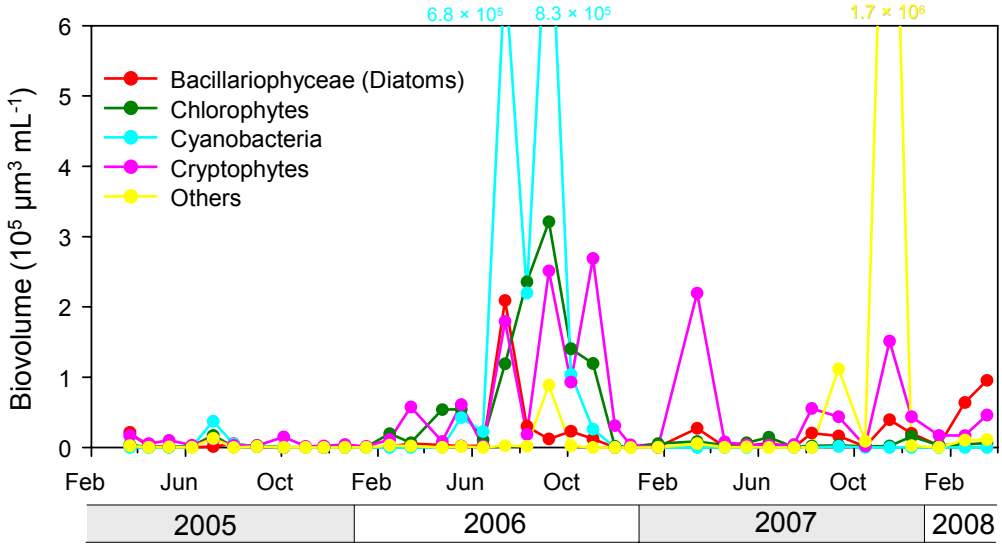


Figure 3.29 Summary of total phytoplankton group biovolumes in Lake I1 (Church Pond), March 2005 to March 2008. 'Bacillariophyceae' refers to all diatoms, and 'others' comprises dinophytes and chrysophytes.

The diatom community of Lake I1 was dominated by centric taxa (Figure 3.30a-e). All diatoms were more abundant from summer 2006 to the end of the monitoring period. Diatoms were dominant during spring and summer months but also present throughout the year. In June 2006 centric diatoms were particularly abundant (Figure 3.30a) reaching a biovolume of 2 x 10^5 µm^3 mL^{-1}, and throughout 2007 centric species were more abundant than in previous years. Pennate diatoms (Figure 3.30b) followed a similar pattern of abundance, becoming more abundant after maximum biovolumes occurred in June 2006. However, the biovolume of pennate species was less than that of centric species. From January to March 2008 all diatom species were more abundant than in previous years. *Asterionella* sp. (Figure 3.30c) was a rare member of the diatom community in Lake I1 until February 2008, reaching a biovolume of 1.2 x 10^4 µm^3 mL^{-1}. *Aulacoseira* spp. (Figure 3.30d) were absent from diatom community throughout the monitoring period, whilst *Synedra* spp. (Figure 3.30e) were only present in significant biovolumes during summer 2006.

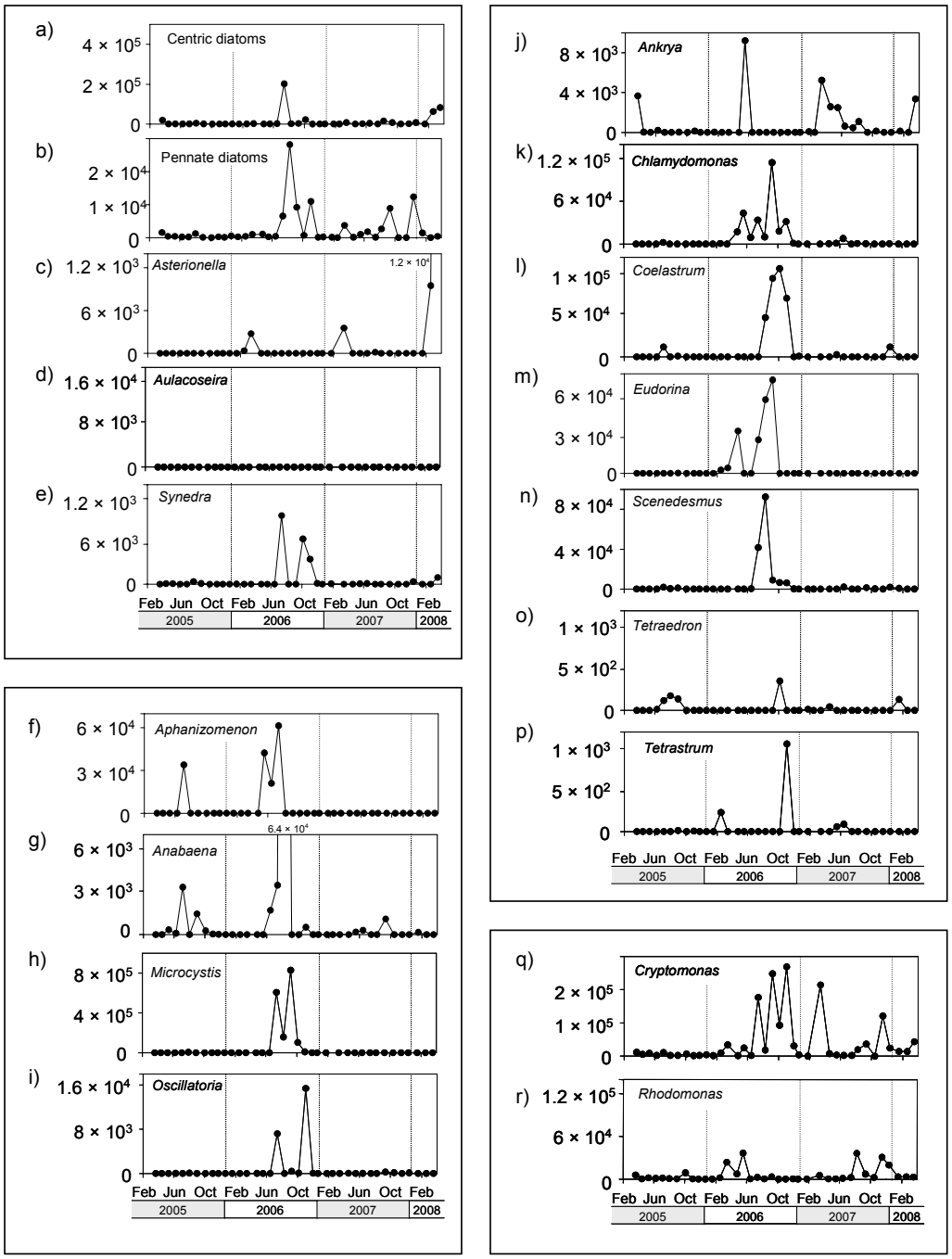


Figure 3.30 Timeseries of biovolumes (µm^3 mL^{-1}) of selected phytoplankton species in Lake I1 (Church Pond), March 2005 - March 2008: a-e) Bacillariophyceae (diatoms); f-i) cyanobacteria; j-p) Chlorophyceae; q-r) Cryptophyceae. Note the differences in *y*-axis scales.

Aphanizomenon flos-aque was the most abundant member of the cyanobacteria community in 2005, although it was only observed during July (Figure 3.30c). *Anabaena* spp. and *Microcystis* sp. were also present although in comparatively smaller biovolumes (Figure 3.30g – h). All cyanobacteria were more abundant during

the summer of 2006, when chlorophyll-*a* concentrations were significantly higher than in 2005 and 2007 (Figure 3.28). *Microcystis* sp. was the most abundant taxon among the cyanobacteria during 2006. *A. flos-aque* and *Anabaena* spp. were present during the early summer and were succeeded by *Microcystis* sp. from July, which remained a member of the phytoplankton community until November 2006. *Oscillatoria* sp. (Figure 3.30i) was only present in significant biovolumes during June and November 2006. The cyanobacteria community during 2007 was relatively depauperate compared to 2006. *Microcystis* sp. was present in small biovolumes during early summer, with *Anabaena* and *Oscillatoria* sp. both present in September. Subsequently no cyanobacterial taxa contributed significantly to phytoplankton biomass until the end of the monitoring period.

Chlorophytes were more abundant during summer months (Figure 3.30j – p) . During summer 2005, the most abundant chlorophyte taxa were *Chlamydomonas* spp. (Figure 3.30k), *Coelastrum* spp. (Figure 3.30l) and *Scenedesmus* spp. (Figure 3.30n, principally *S. opoliensis*). *Pediastrum duplex* and *Tetraedron triangulare* (Figure 3.30o) were also present. During 2006, chlorophytes were more abundant than in 2005 and more species were represented in the community. An increase in the abundance of *Eudorina elegans*, *Coelastrum* spp., and *Scenedesmus* spp. coincided with a large increase in chlorophyll-*a* concentration during the summer of 2006 (Figure 3.28). *Chlamydomonas* spp. were also present during the summer of 2006. *S. communis, S. falcatus* and *S. opoliensis* were present during summer 2006. *Staurastrum* spp., *P. boryanum*, *Monoraphidium* spp. and *Micratinium* spp. were also present in the chlorophyte community during 2006. Chlorophytes were rare during 2007 compared to 2006. Small species such as *A. judyaii* and *Chlamydomonas* spp. and *S. communis* were observed from March to July 2007. *A. judyaii, Monoraphidium* spp. and *Schroderia* spp. increased in abundance from January 2008 to March 2008.

Cryptomonas spp. and *Rhodomonas* spp. were present throughout the monitoring period in Lake I1 (Figure 3.30q-r). During 2005, *Cryptomonas* spp. was present in relatively low biovolumes compared to the rest of the monitoring period, rarely exceeding 20,000 $\mu m^3 mL^{-1}$. Large biovolumes, in excess of 200,000 $\mu m^3 mL^{-1}$ occurred during late 2006 and March 2007. Unlike many other phytoplankton species, *Cryptomonas* spp. were present throughout 2007, with significant biovolumes

recorded during November 2007. *Rhodomonas* spp. were comparatively less abundant than *Cryptomonas* spp. but still formed a substantial proportion of the phytoplankton community. The highest biovolume of *Rhodomonas* spp. in 2005 occurred in October, and was again abundant during spring and early summer of 2006, reaching a biovolume of 36,300 μm^3 mL^{-1} in May 2006. During early 2007, *Rhodomonas* spp. was not present in biovolumes comparable to early 2006, although it was a significant member of the phytoplankton community during autumn and winter 2007.

Trachelomonas sp. were observed in spring and early summer during 2005 and 2006. In 2007, *Trachelomonas* sp. were abundant from October to December, forming a significant proportion of the winter phytoplankton community. *Trachelomonas* sp. were again observed during February and March 2008. *Phacus* sp. and *Euglena* spp. were never abundant members of the phytoplankton community, both reaching peak biovolumes during the summer of 2006 (~3000 μm^3 mL^{-1}). The only dinophytes observed were *Peridinium* spp. which was recorded during the summers of 2005 and 2006, and *Dinobryon cylindricum*, observed in September 2007 only.

3.5.3 Macrophytes

Submerged macrophytes were abundant in Lake I1 (Figure 3.31a-c). *Elodea canadensis* was the dominant species of the submerged macrophyte community in all years. During 2005, *Chara* spp. were abundant, principally *C. contraria*, although *C. globularis* and *C. vulgaris* var. *papillata* were also recorded. Only *C. contraria* was found during 2006 and 2007. *Potamogeton* spp. were abundant during 2005 and 2007, including *P. trichoides* and *P. pectinatus* in 2005 and *P. crispus* and *P. pectinatus* during 2007. Only *P. pectinatus* was recorded during 2006. *C. hermaphroditica* was frequently found during 2005, although restricted to only five and four sampling sites during 2006 and 2007 respectively. *Hydrodictyon* sp. was only recorded during 2005. Filamentous algae were found during all three years.

In all years, the dominant emergent macrophyte species recorded were *G. maxima* and *S. erectum*. *T. latifolia* was found in all years, mainly along the north-eastern shore of the lake although was considerably rarer. *Carex* sp. and *Juncus* sp. were

observed during 2005 and 2006, although *Carex* sp. was not found in 2007. *Acorus calamus* was recorded during 2006 and 2007. *Iris pseudacorus* and *Schoenoplectus. lacustris* were both observed in 2005 and 2007 only. The only floating-leaved macrophyte observed in Lake I1 was *Polygonum amphbium*, in 2006 and 2007.

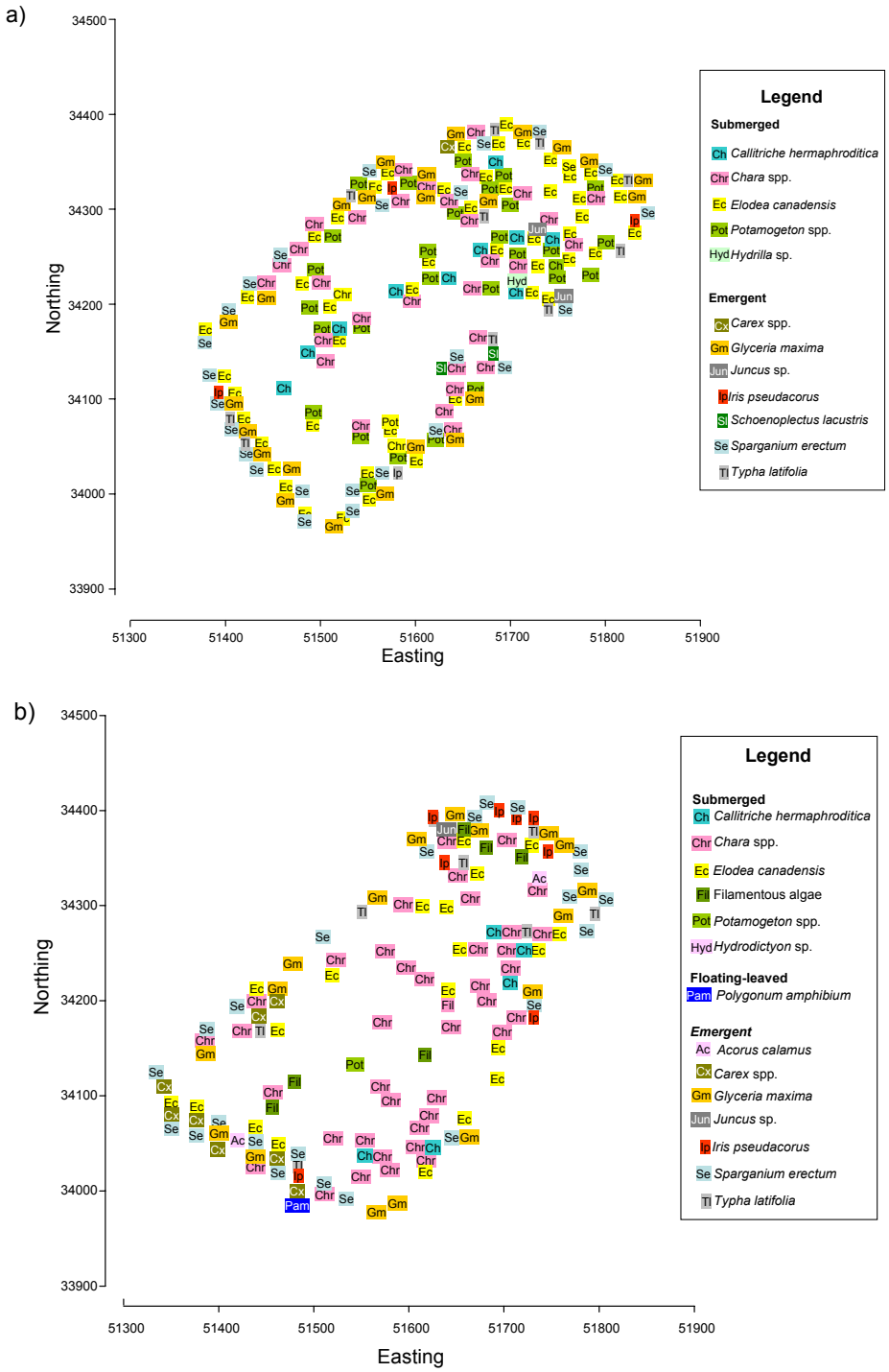


Figure 3.31 Schematic maps of the occurrence of aquatic macrophytes observed in Lake I1 (Church Pond) during a) 2005; b) 2006 and c) 2007 (overleaf).

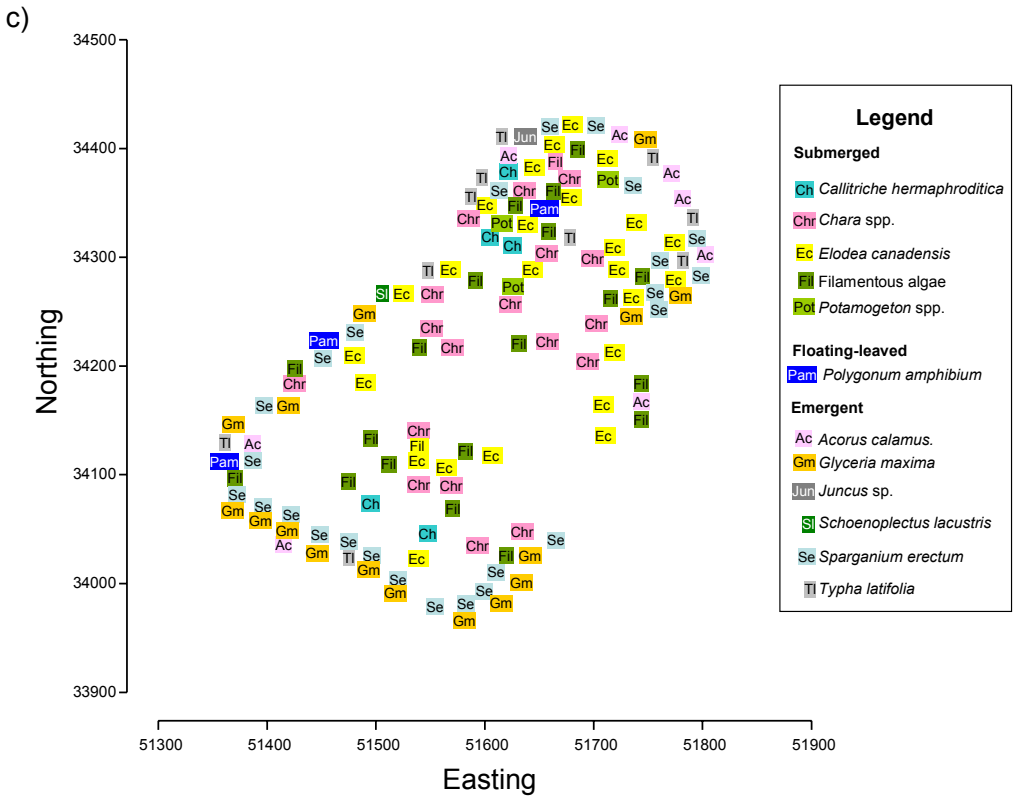


Figure 3.31 contd.

3.5.4 Zooplankton

Total zooplankton densities tended to be largest during late summer and winter. Throughout the monitoring period, the total number of zooplankton was less than 10 ind. L^{-1}, except during September and November 2006, and again during March 2005 when a large peak in total zooplankton biomass (78 ind. L^{-1}) occurred, predominantly cyclopoid copepods and *B. longirostris*.

The zooplankton community in Lake I1 was dominated by *D. hyalina*, calanoid copepods and cyclopoid copepods (Figure 3.32). Calanoid copepods were generally less abundant than cyclopoid species. The maximum abundance of cyclopoid copepods was observed during September 2006 of over 40 ind. L^{-1}. The maximum

abundance of calanoid copepods was 3 ind. L^{-1}, during summer 2005 and 2007. *B. longirostris* was only present in significant densities (maximum 25 ind. L^{-1}) during October and November 2006. The rotifer *Asplancha* sp. was recorded during autumn 2006 and spring 2007, reaching a maximum of 6 ind. L^{-1}. *D. curvirostris* was observed during January and February 2008 although in low densities (<1 ind. L^{-1}). *Ceriodaphnia* sp. was present in the zooplankton community during the winters of 2005 and 2006, and during the late summer and autumn of 2007, although in low densities. *Sida* sp. was also occasionally observed during the summer of 2007.

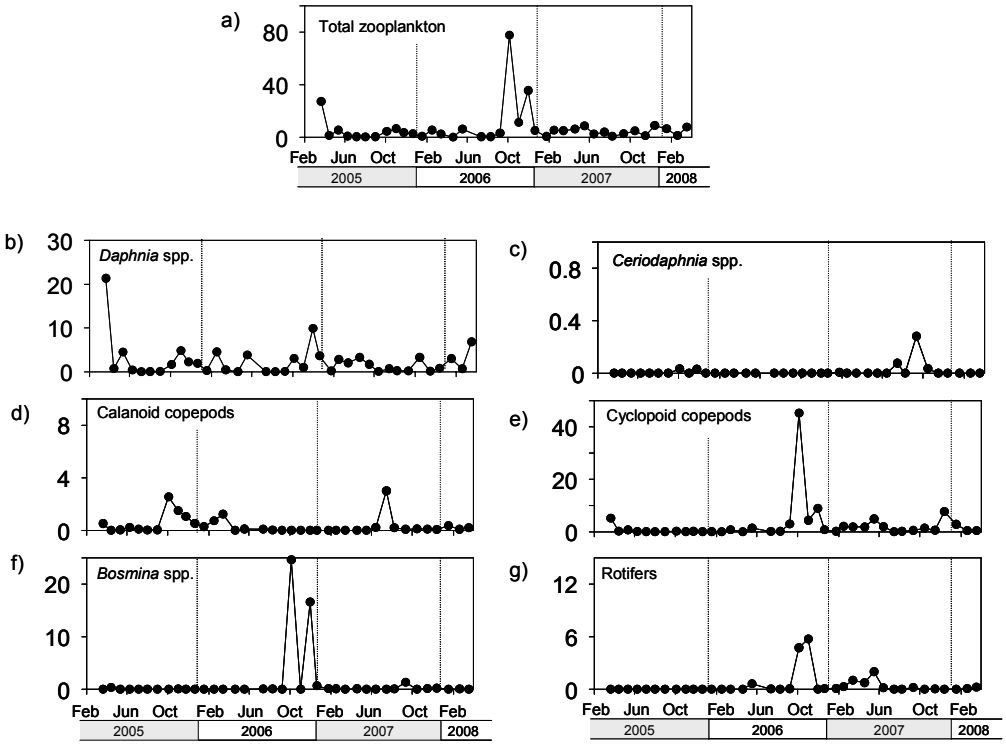


Figure 3.32 Timeseries of zooplankton abundance: a) total abundance; b-g) selected groups, March 2005-March 2008 in Lake I1 (Church Pond). All data are presented as individuals L^{-1}.

3.6 Lake I2 (Clifton Pond)

Figure 3.33 Lake I2 (Clifton Pond) looking approximately north, September 2007. Lake I2 is isolated from the River Erewash. Artificially-established reedbeds of *P. communis* can be seen on the left of the photograph.

3.6.1 Water chemistry

The mean TP in Lake I2 (Figure 3.33) was 73 (±7) µg L^{-1} and ranged between 3 and 199 µg L^{-1} during the monitoring period (Figure 3.34a). TP concentrations in 2005 and 2006 increased during spring and in both years decreased briefly between May and June, before increasing rapidly throughout the summer, reaching a maximum in August. Concentrations then fell throughout the autumn and winter, to less than 10 µg L^{-1} in March 2006 and May 2007. After May 2007, TP concentrations increased abruptly to 115 µg L^{-1} and fell gradually until December 2007 (73 µg L^{-1}) before increasing to 84 µg L^{-1} in March 2008. Mean SRP during the monitoring period was 26 (±4) µg L^{-1}, ranging between <1 and 91 µg L^{-1} (Figure 3.34a). The highest SRP concentrations were recorded during September and October with the maximum concentrations each year declining throughout the monitoring period, from 91 µg L^{-1} in 2005 to 33 µg L^{-1} in 2007. SRP concentrations typically increased from spring to late

summer, with a break in this trend during May to June (except for 2007), after which SRP concentrations rose rapidly until the end of summer. Concentrations of SRP decreased throughout the winter to reach <10 µg L^{-1} during winter.

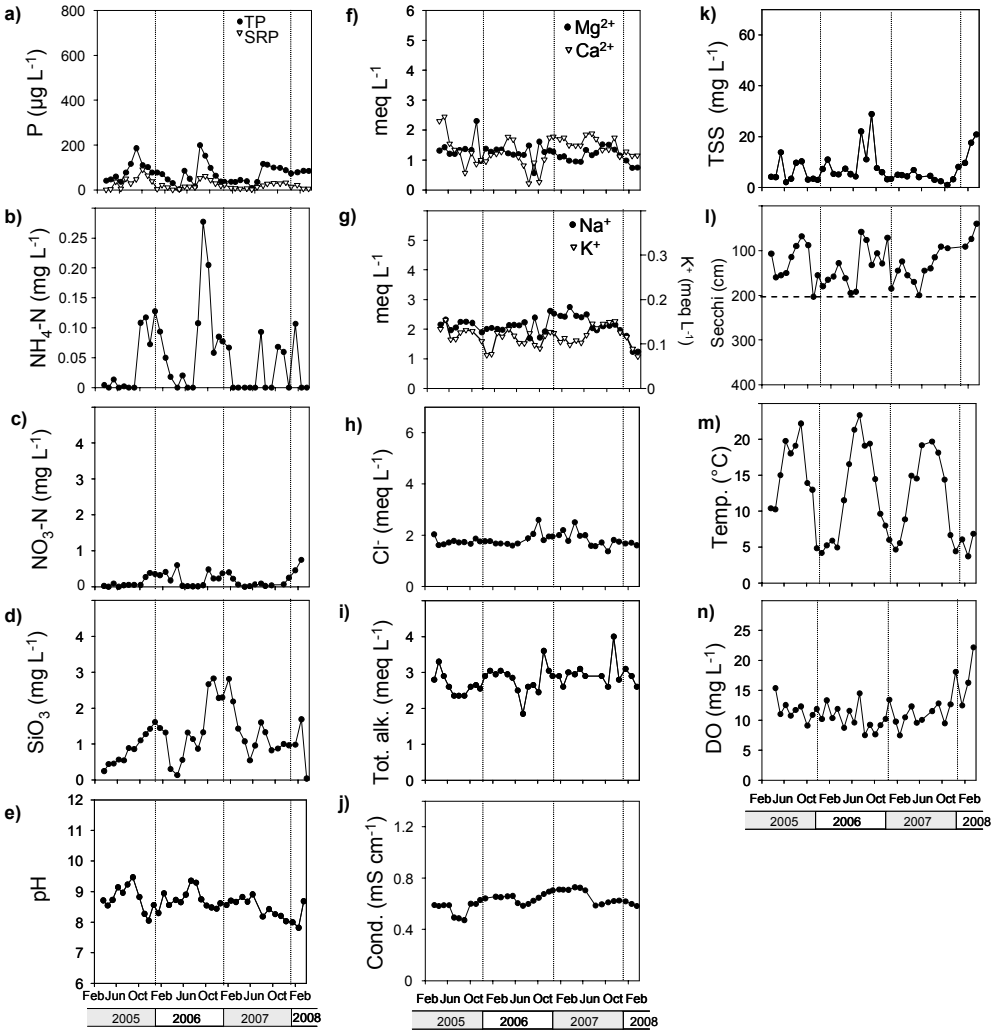


Figure 3.34 Water chemistry and physico-chemical properties of Lake I2 (Clifton Pond) March 2005 to March 2008: a) TP and SRP (µg L^{-1}); b) NH_4-N (mg L^{-1}); c) NO_3-N (mg L^{-1}); d) SiO_3 (mg L^{-1}); e) pH; f) Mg^{2+} and Ca^{2+} (meq L^{-1}); g) Na^+ and K^+ (meq L^{-1}); h) Cl^- (meq L^{-1}); i) total alkalinity (meq L^{-1}); j) specific conductivity (mS cm^{-1}); k) TSS (mg L^{-1}); l) Secchi disk depth (cm, dotted line represents approximate lake depth); m) surface temperature (°C); n) DO (mg L^{-1}).

NH_4-N concentrations (Figure 3.34b) were highly variable ranging between 0 and 0.28 mg L^{-1} (mean, 0.05 ±0.01 mg L^{-1}) during the monitoring period. NH_4-N concentrations were highest during the winter in 2005 and 2006, typically increasing from below detection limits in July and August to 0.13 mg L^{-1} in December 2005 and 0.28 mg L^{-1} in September 2006. NH_4-N concentrations decreased rapidly from wintertime maxima to

<0.02 mg L^{-1} during spring. In 2007, large wintertime increases were not evident, and NH_4-N oscillated throughout the summer between 0 and > 0.06 mg L^{-1}. NO_3-N concentrations (Figure 3.34c) were low throughout the monitoring period: the mean NO_3-N concentration was 0.2 (±0.03) mg L^{-1}, with a range of 0-0.8 mg L^{-1}. Throughout the summer of all years, NO_3-N concentrations were often undetectable and were followed by steep rises during November to reach 0.4 mg L^{-1} in 2005 and 0.5 mg L^{-1} in 2006. Elevated concentrations were maintained throughout the summer before sharp falls in early summer. After November 2008, NO_3-N concentrations rose rapidly from <0.1 mg L^{-1} to over 0.7 mg L^{-1} in March 2008.

SiO_3 concentrations (Figure 3.34d) increased steadily from 0.2 mg L^{-1} in March 2005 to 1.6 mg L^{-1} by December 2005, then decreased rapidly to 0.1 mg L^{-1} in April 2006. SiO_3 concentrations rose rapidly throughout summer and autumn 2006 to reach 2.8 mg L^{-1} in November 2006, and then decreased throughout January to May 2007 to 0.5 mg L^{-1}. SiO_3 concentrations were lower during the winter of 2007 than 2005 and 2006, and decreased sharply from February to March 2008.

pH in Lake I2 (Figure 3.34e) ranged between 7.48 and 9.79 with a mean of 8.67 (±0.09) throughout the monitoring period. pH showed a decreasing trend during the monitoring period (r_s = -0.466, p = 0.003). During 2005 and 2006, the highest pH occurred during July and August and the lowest in November. In 2007, the highest pH was recorded in June (8.91) and the lowest in December (8.18). pH fell from September 2007 to February 2008 before increasing sharply to 8.69 in March 2008. pH did not change substantially throughout the water column (Figure 3.35a) during the monitoring period.

Mg^{2+} concentrations in Lake I2 (Figure 3.34f) ranged between 0.6 and 2.3 meq L^{-1} with a mean of 1.34 (± 0.05) meq L^{-1}. Mg^{2+} concentrations throughout 2005 and 2006 did not show any seasonality. During 2007, Mg^{2+} concentrations decreased from January throughout the spring to 0.9 meq L^{-1} in May 2007, before increasing to 1.5 meq L^{-1} during the autumn. Mg^{2+} concentrations decreased progressively to 0.7 meq L^{-1} by March 2008. Ca^{2+} concentrations (Figure 3.34f) ranged between 0.2 and 2.5 meq L^{-1} and averaged 1.3 (±0.08) meq L^{-1}. Concentrations of Ca^{2+} were higher during winter and spring in 2005 and 2006, and decreased rapidly from April to August

each year. Unlike 2005 and 2006, during 2007, Ca^{2+} concentrations fell only slightly throughout the year.

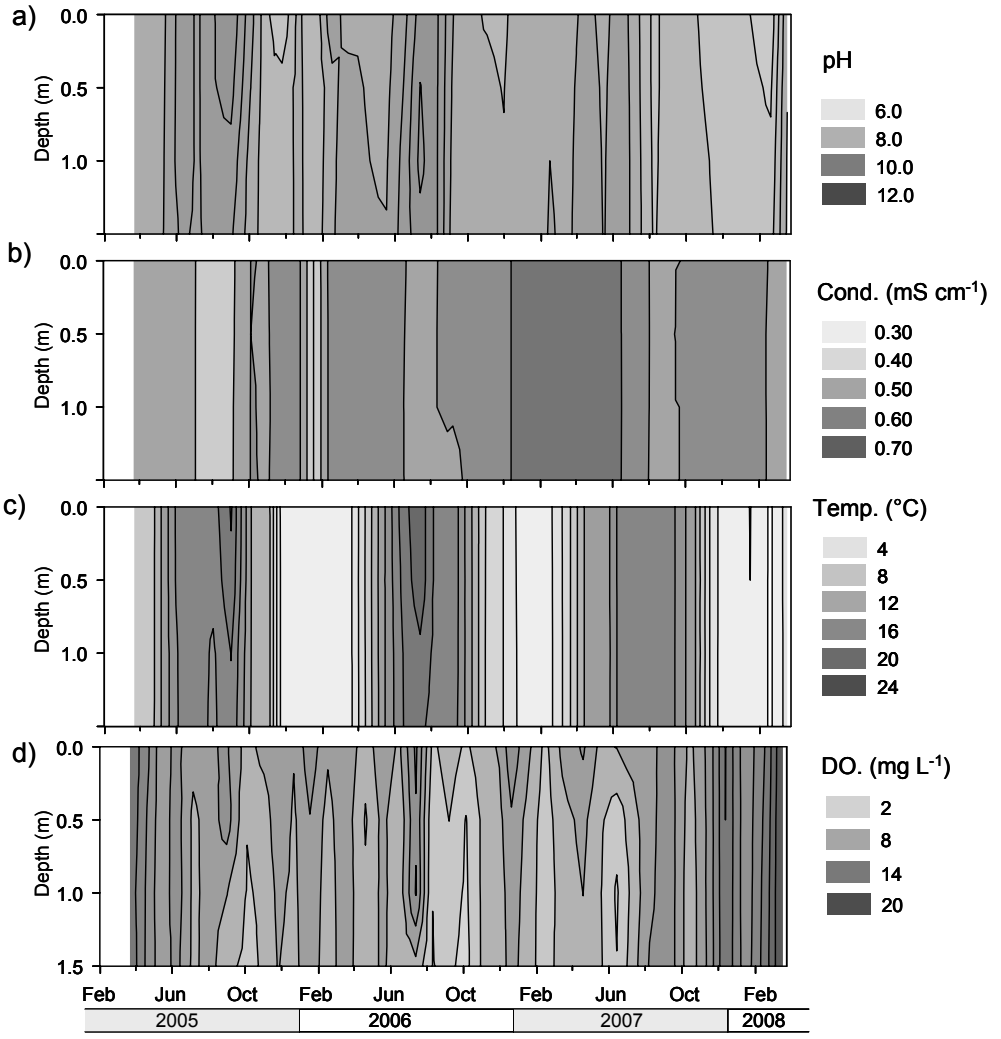


Figure 3.35 Profiles of a) pH; b) specific conductivity (mS cm^{-1}); c) temperature (°C); d) dissolved oxygen (mg L^{-1}) in Lake I2 (Clifton Pond), March 2005 to March 2008. *y*-axis denotes depth below water surface (m).

The mean Na^+ concentration during the monitoring period was 2.1 (±0.5) meq L^{-1} and ranged between 1.2 and 2.8 meq L^{-1} (Figure 3.34g). During 2005 and summer 2006, Na^+ concentrations remained between 1.9 and 2.3 meq L^{-1}, and then increased sharply between November and December 2006 to ~2.5 meq L^{-1}. From July 2007 to March 2008 Na^+ concentrations fell, reaching 1.2 meq L^{-1} by the end of the monitoring period. No seasonality was observed in K^+ concentrations (Figure 3.34g) which ranged between 0.07 and 0.51 meq L^{-1}. (mean, 0.1 ± 0.003 meq L^{-1}). A sharp

decrease in K^+ concentrations was observed between November 2007 and March 2008 from 0.15 to 0.07 meq L^{-1}. Cl^- concentrations (mean, 1.8 ±0.04 meq L^{-1}; range, 1.4–2.6 meq L^{-1}, Figure 3.34h) remained relatively constant throughout March 2005 to July 2006, increased during the winter of 2006, reaching a maximum of 2.6 meq L^{-1} during October 2006. Cl^- concentrations remained above 1.8 meq L^{-1} until May, until falling until the end of the monitoring period. Mean total alkalinity (Figure 3.34i) during the monitoring period was 2.8 (±0.06) meq L^{-1} and ranged between 1.85 and 4.00 meq L^{-1}. No clear seasonality was evident.

Mean specific conductivity in Lake I2 was 0.64 (±0.02) mS cm^{-1} and ranged between 0.40 and 1.16 mS cm^{-1} during the monitoring period (Figure 3.34j) Specific conductivity decreased during the summers of 2005 and 2006, and tended to increase throughout autumn and winter. Specific conductivity declined sharply from 0.70 mS cm^{-1} in June 2007 to 0.59 in August 2007, and remained between 0.58 and 0.63 mS cm^{-1} until March 2008. Specific conductivity did not change with depth during the monitoring period (Figure 3.35b).

TSS concentrations (Figure 3.34k) ranged between 0.9 and 28.8 mg L^{-1} during the monitoring period. The mean TSS concentration was 7.3 (±1.0) mg L^{-1}. From March 2005 to June 2006, TSS concentrations varied between 2 and 13.8 mg L^{-1}, with maximum concentrations occurring during May and August 2005 and January 2006. TSS concentrations during the summer of 2006 were considerably higher than either 2005 or 2007, peaking at 28.8 mg L^{-1} in August 2007, before falling to <7 mg L^{-1} until December 2007 after which TSS concentrations increased rapidly to 20.8 mg L^{-1} in March 2008.

Secchi disk depths (Figure 3.34l) in Lake I2 frequently exceeded the depth of the lake (~2 m) due to the shallowness of the lake. The lowest Secchi disk depth was 40 cm, recorded in March 2008. During 2005, Secchi disk depths decreased throughout the summer to 68 cm in August, before rapidly increasing to over 200 cm in November. Secchi disk depths remained high throughout winter 2005, before decreasing rapidly to 58 cm in July. Secchi disk depths increased between August 2006 and June 2005, after which Secchi disk depths progressively decreased to the end of the monitoring period.

Surface water temperature (Figure 3.34m) was strongly seasonal. Mean surface temperature was 12.2°C and ranged between 3.7 and 23.4°C throughout the monitoring period. Highest temperatures occurred in July or August each year, and in 2007 the highest temperature was 19.7°C, compared to 22.2°C and 23.6°C in 2005 and 2006. For much of the monitoring period, water temperature was homogenous throughout the water column (Figure 3.35c), although during the summers of 2005 and 2006 the surface water was 2-3°C warmer that at 1.5 m depth.

DO concentrations (Figure 3.34n) ranged between 7.5 and 22.2 mg L^{-1} throughout the monitoring period; the mean DO concentration was 11.7 (±0.5) mg L^{-1}. Throughout 2005 and 2006, DO concentrations were variable, with no seasonal trends apparent. From early summer 2007, DO concentrations showed a gradual increase towards the end of the monitoring period. DO concentrations were generally higher nearer the surface of the lake than the bottom (Figure 3.35), by 4-5 mg L^{-1}. From summer 2007 to the end of the monitoring period no gradients of DO concentrations were observed.

3.6.2 Phytoplankton

Chlorophyll-*a* concentrations

Chlorophyll-*a* concentrations (Figure 3.36) were highly seasonal in Lake I2. From concentrations of <10 µg L^{-1} through spring and early summer, rapid increases were observed during 2005 and 2006, reaching 30 µg L^{-1} in August 2005 and 81 µg L^{-1} during July 2007. Abrupt decreases in chlorophyll-*a* concentrations followed summertime peaks. During 2007, no late summer increases were observed, with the maximum concentration being recorded in December (34 µg L^{-1}). In 2008, chlorophyll-*a* concentrations rose sharply from January to 61 µg L^{-1} in March. Throughout the monitoring period, the mean chlorophyll-*a* concentration was 13 (±3) µg L^{-1}, and ranged between 2 and 81 µg L^{-1}.

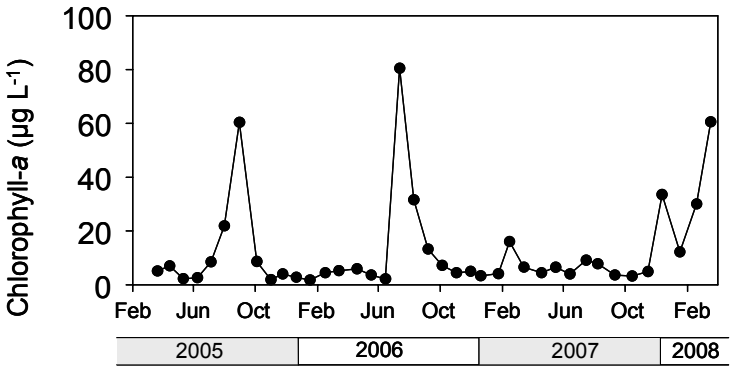


Figure 3.36 Chlorophyll-*a* concentrations in Lake I2 (Clifton Pond), March 2005 to March 2008.

Community composition

The phytoplankton community of Lake I2 was generally dominated by cryptophytes, chlorophytes and cyanobacteria (Figure 3.37). During the summer of 2005, cryptophytes and chlorophytes were more abundant than other phytoplankton groups, and in 2006 cyanobacteria also formed a significant component of the phytoplankton community. In summer 2007, diatoms were the most abundant group of phytoplankton and cryptophytes, chlorophytes and cyanobacteria were rare compared to 2005 and 2006. From winter 2007, to the end of the monitoring period, all phytoplankton groups except cyanobacteria became substantially more abundant than in 2005 or 2006. Diatoms were abundant during autumn 2007 and cryptophytes and chlorophytes became the most abundant phytoplankton groups by the end of the monitoring period.

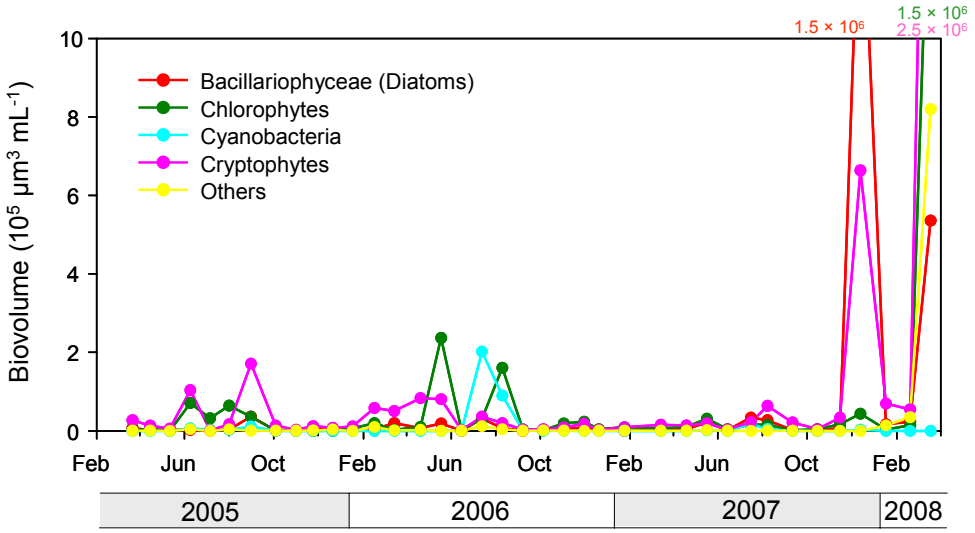


Figure 3.37 Summary of total phytoplankton group biovolumes in Lake I2 (Clifton Pond), March 2005 to March 2008. 'Bacillariophyceae' refers to all diatoms, and 'others' comprises dinophytes and chrysophytes.

Diatoms were generally scarce through most of the monitoring period in Lake I2. Centric diatoms (Figure 3.38a), although present in most samples, did not contribute significantly to the total biomass of phytoplankton. They were most abundant during August and September in 2005 (3.6×10^4 µm^3 mL^{-1}) and 2006 (2.1×10^4 µm^3 mL^{-1}). During 2007, centric diatoms were less abundant during summer (1.5×10^4 µm^3 mL^{-1}), although a substantial increase in biovolume occurred in December 2007, when 1.5×10^6 µm mL^{-1} of centric species was observed. Although scarce during January and February 2008, centric diatoms were again abundant during March (3.6×10^5 µm^3 mL^{-1}). Pennate diatoms (Figure 3.38b) were less abundant during the monitoring period than centric species, generally recorded only in small biovolumes. During 2007, pennate species became more abundant in phytoplankton samples, reaching 6×10^3 µm^3 mL^{-1} in August, and increased rapidly between February and March 2008 to 2.4×10^4 µm^3 mL^{-1}. *Asterionella* sp. (Figure 3.38) was observed occasionally during spring and early summer in low biovolumes. *Aulacoseira* spp. (Figure 3.38d) were first recorded during the summer of 2006 (8×10^3 µm^3 mL^{-1} in August 2006), and was present in late summer and autumn 2007, although did not contribute significantly to the biomass of the diatom community. *Synedra* spp. (Figure 3.38e) were also rare in Lake I2, observed in substantial biovolumes in early summer 2006 and March 2008. *Meridion* sp. was recorded in low biovolumes throughout the monitoring period although only contributed substantially to the biovolume of the diatom community in March 2008 (3.2×10^3 µm^3 mL^{-1}).

Anabaena sp. (Figure 3.38f) was the most frequently recorded member of the cyanobacteria community in Lake I2. During 2005, *Anabaena* sp. was present at up to 2.7×10^3 µm^3 mL^{-1}, and in July 2006 was abundant, reaching 2.0×10^4 µm^3 mL^{-1}. Although absent throughout most of the monitoring period, *Oscillatoria* sp. was also abundant in July 2007 (4.1×10^4 µm^3 mL^{-1}). *Aphanizomenon* sp. was visible in the water column during summer 2006, although the species was not present in the sample counted, as colonies had aggregated at the surface, suggesting that the data shown in Figure 3.38g probably underestimate the abundance of *Aphanizomenon*.

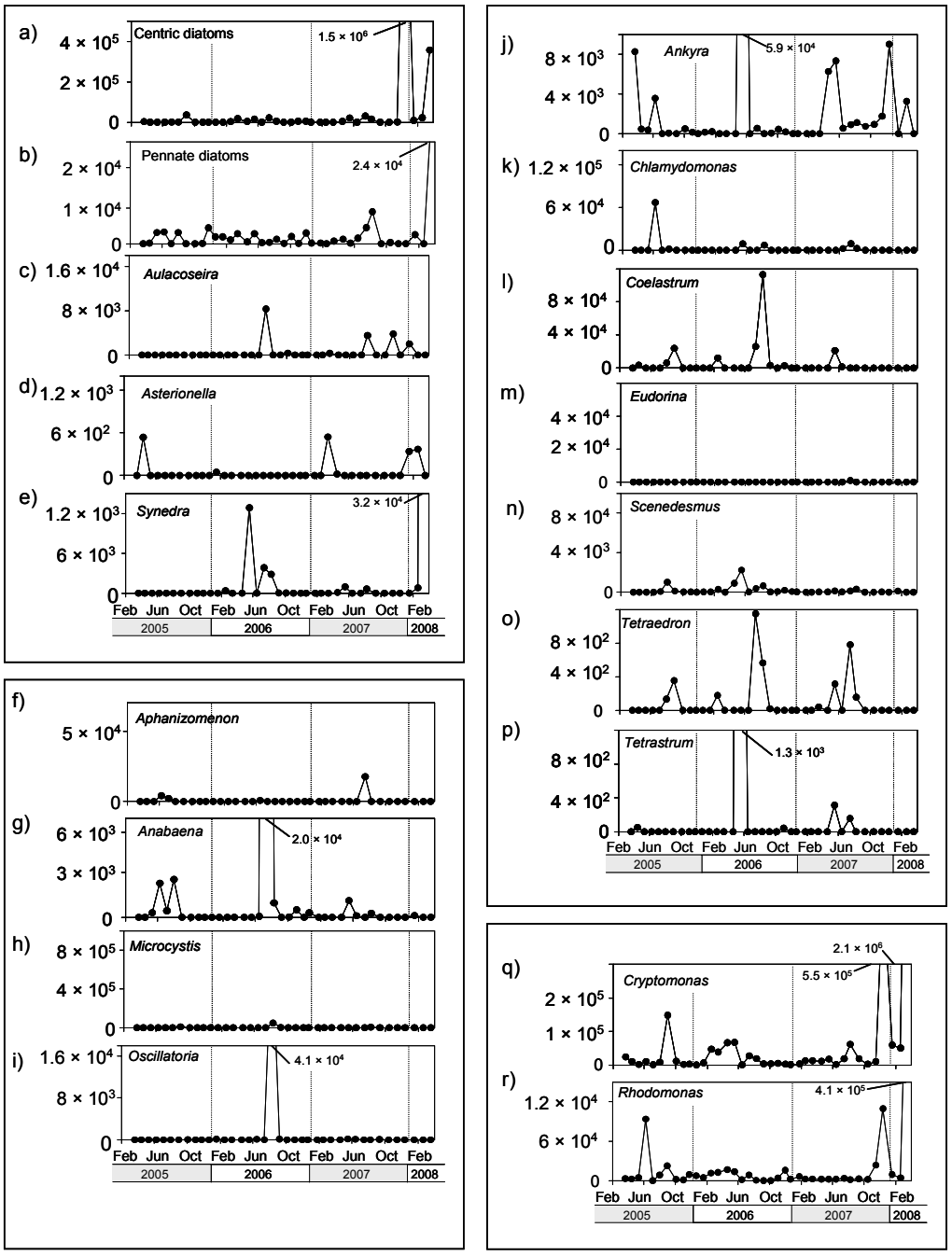


Figure 3.38 Timeseries of biovolumes (µm^3 mL^{-1}) of selected phytoplankton species in Lake I2 (Clifton Pond), March 2005-March 2008: a-e) Bacillariophyceae (diatoms); f-g) cyanobacteria; j-p) Chlorophyceae; q-r) Cryptophyceae. Note the change in *y*-axis scales.

Chlorophyte species were generally present in smaller biovolumes than other groups in Lake I2. Although observed throughout the monitoring period, *A. judyaii* (Figure 3.38j) did not contribute significantly to the total phytoplankton biomass. *A. judyaii* was most frequently recorded during May and June, however the species was also observed during March 2005 and December 2008 in relatively high biovolumes. *Chlamydomonas* spp. (Figure 3.38k) were one of the most abundant chlorophytes recorded in Lake I2. The highest biovolume of *Chlamydomonas* spp. occurred during June 2005 (6.7×10^4 µm^3 mL^{-1}) and were also present in the phytoplankton community during early summer 2006 and 2007, although at substantially lower biovolumes ($< 9 \times 10^3$ µm^3 mL^{-1}) than 2005. *Coelastrum* sp. (Figure 3.38l) was also present during summer months in high biovolumes compared to other chlorophytes, exceeding 1.1×10^5 µm mL^{-1} in August 2006. In 2005 and 2007, the biovolume of *Coelastrum* sp. reached ~2.1×10^4 µm^3 mL^{-1}. *Eudorina* sp. (Figure 3.38m) was absent in the phytoplankton community throughout the monitoring period except for an isolated occurrence during July 2007. *Scenedesmus* spp. (Figure 3.38n) were most abundant during summer 2006 although contributed a relatively small biovolume to the phytoplankton community. *S. communis* and *S. opoliensis* were the most frequently observed members of the genus. *Scenedesmus* spp. were rare throughout both 2005 and 2007. *Tetrastrum* spp. (Figure 3.38o) and *Tetraedron* spp. (Figure 3.38p) were both rare throughout the monitoring period, rarely exceeding 8×10^2 µm mL^{-1}, except for May 2006 when 1.3×10^3 µm^3 mL^{-1} of *Tetrastrum* was recorded. *T. caudatum* and *T. triangulare* were the most abundant members of the *Tetraedron* genus. Other chlorophytes observed during the monitoring include *Dictyosphaerium* sp. which was present in relatively large biovolumes (>2.7×10^3 µm^3 mL^{-1}) during summer 2005 and 2006, although not observed during 2007. *Monoraphidium* sp. was observed during the summer of 2005 and 2006, and was present in a large biovolume (4.3×10^5 µm^3 mL^{-1}) during March 2008. Small green flagellates also became more dominant from October 2007 to March 2008, comprising 1×10^6 µm^3 mL^{-1} in March 2008.

Cryptomonas spp. (Figure 3.38q) were present throughout the monitoring period. In 2005, the highest abundance (1.4×10^5 µm^3 mL^{-1}) of *Cryptomonas* spp. occurred during August, and in 2006 a maximum biovolume of 6.6×10^4 µm^3 mL^{-1} occurred during May. During 2007, the biovolume of *Cryptomonas* spp. reached 5.5×10^5 µm^3

mL^{-1} during December, fell to ~5.5 × 10^4 μm^3 mL^{-1} during January and February 2008 and then increased by an order of magnitude to 2.1 × 10^6 μm^3 mL^{-1} in March 2007. *Rhodomonas* spp. (Figure 3.38r) was also common during the monitoring period. During 2005, the biovolume of *Rhodomonas* spp. reached 9.3 × 10^4 μm^3 mL^{-1} in June and continued to be present in the phytoplankton community throughout winter 2005 and spring 2006. The biovolume of *Rhodomonas* spp. were generally smaller in 2006 than 2005, peaking at only 1.7 × 10^4 μm^3 mL^{-1} in May. *Rhodomonas* spp. were present in small quantities throughout the summer of 2006 and during 2007. From October 2007 to December 2007 the biovolume of *Rhodomonas* spp. increased sharply from 1.7 × 10^3 to 1.1 × 10^5 μm^3 mL^{-1}, and then decreased during January and February to 5.0 × 10^4 μm^3 mL^{-1} before increasing sharply to 4.1 × 10^5 μm^3 mL^{-1} in March 2008.

Dinophytes were generally rare throughout the monitoring period in Lake I2. Both *Gymnodinium* spp. and *Peridinium* spp. were observed occasionally, generally during summer in small biovolumes (<5 × 10^3) μm^3 mL^{-1}, except for March 2008 when *Gymnodinium* spp. reached 4.9 × 10^5 μm^3 mL^{-1}. *Ceratium hirundella* was found in Lake I2 during July 2006 although was not present otherwise. Euglenophytes were also rare throughout the monitoring period, although a large increase in the biovolume of *Trachelomonas* sp. was observed from January 2008 (5 × 10^3 μm^3 mL^{-1}) to March 2008 (2.4×10^5 μm^3 mL^{-1}). *Phacus* sp. was rarely observed during the monitoring period, only present in significant biovolumes in March 2008 (8.9 × 10^4 μm^3 mL^{-1}).

3.6.3 Macrophytes

Submerged macrophytes (Figure 3.39a-c) were abundant in Lake I2 during 2005 and 2007 although were relatively scarce during 2006. *Elodea canadensis*, *Chara* spp. and *Potamogeton* spp. were the most abundant submerged species during 2005 and 2007. *Chara* spp. were particularly abundant during 2007, although rarely recorded during 2006. The genus *Chara* was represented by *C. vulgaris* in all years, and additionally *C. globularis* during 2005. *Potamogeton* spp. were widespread during 2006, not recorded during 2006 and widespread in 2007. *P. pectinatus*, *P. trichoides* and *P. crispus* were observed during 2005. *P. trichoides* and *P. pectinatus* were recorded during 2007. Filamentous algae, were frequently observed in all years. Less abundant submerged species included *Callitriche hermaphroditica* observed in 2005

and 2007, and *Myriophyllum spicatum*, recorded during 2006 and 2007. *Ranunculus circinatus* was observed in 2005 and 2007 in south-western areas of the lake only. The floating-leaved *Polygonum amphibium* was present in Lake I2 during all years.

Emergent macrophytes were found along all shores of the lake. *Phragmites communis* was frequently encountered along the north-western areas of the lake, where artificial reedbeds have been established (see Figure 3.33, page 146). *P. communis* was more widespread during 2006. *Glyceria maxima* was observed in all years, generally restricted to the south-west of the lake. *Juncus* sp. was found in all years in the west of the lake. *Iris pseudacorus* was found in 2006. *Acorus calamus* was scarce, and only observed during 2005 and 2006. Isolated occurrences of *Typha latifolia* were observed in all years. *Phalaris arundinacea* was found along the southern shore during 2006 and 2007. *Sparganium erectum* was found in all three years although at few sampling sites.

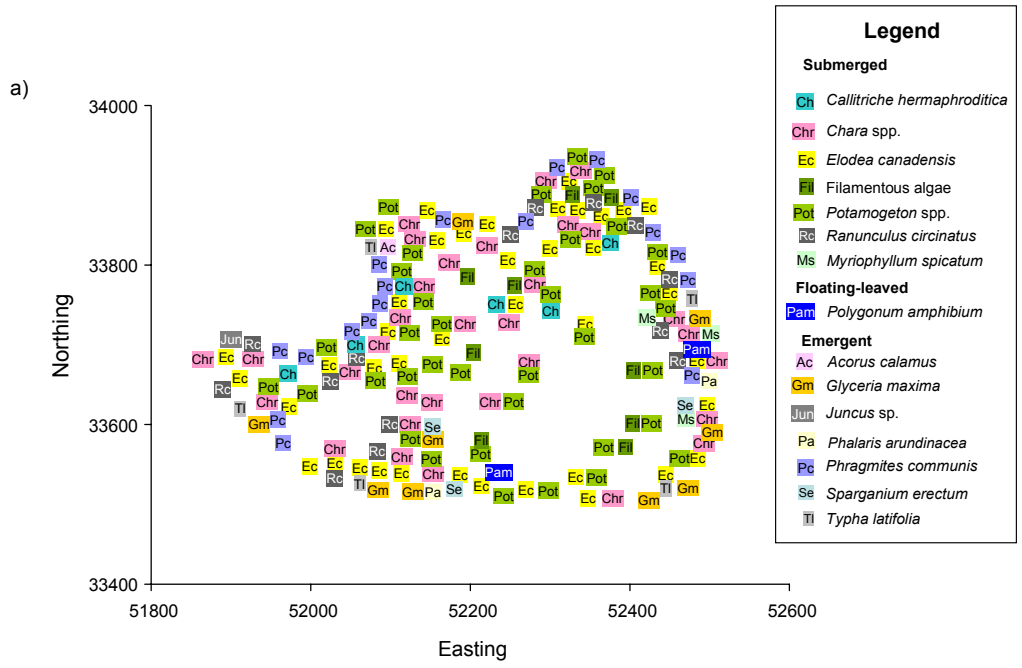
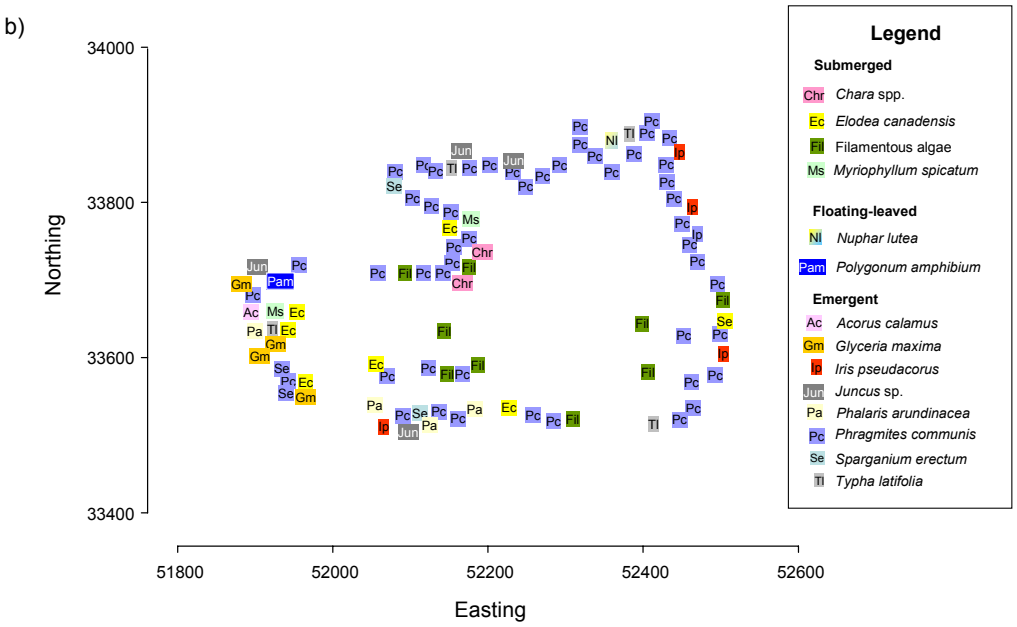
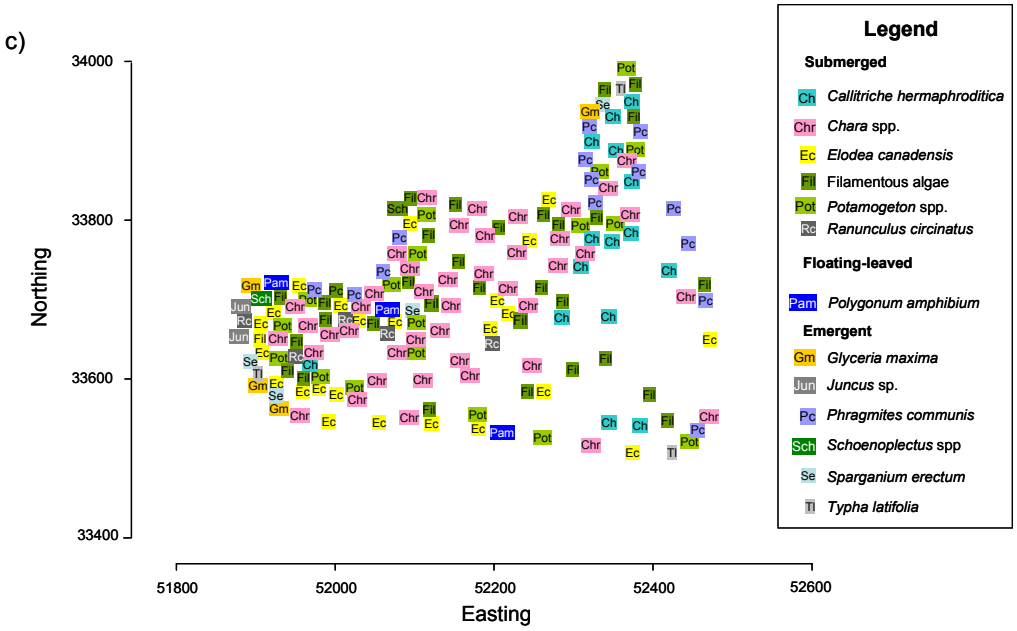


Figure 3.39 Schematic maps of the occurrence of aquatic macrophytes observed in Lake I2 (Clifton Pond) during a) 2005, and (overleaf) b) 2006 and c) 2007.





3.6.4 Zooplankton

Total zooplankton abundance in Lake I2 (Figure 3.40a) increased from < 1 to 18 ind. L^{-1} from August to October 2005, before decreasing gradually throughout the winter. From July to September 2006, total zooplankton abundance increased sharply and then decreased to < 10 ind. L^{-1} during winter 2006. In 2007, total zooplankton abundance briefly increased during spring (to 13 ind. L^{-1}), fell to < 4 ind. L^{-1} during May and June, then rose again in late summer. The total abundance of zooplankton

decreased between August and November and remained at densities of < 3 ind. L^{-1} until March 2008.

Daphnia spp. (Figure 3.40b) were regularly observed in the zooplankton community, including *D. hyalina*, *D. hyalina* var. *lacustris*, *D. pulex*, and *D. cucullata*. *Daphnia* spp. were observed during early summer 2005 at densities up to 3 ind. L^{-1}, and then over 6 ind. L^{-1} in late summer. Densities fell progressively until February 2006 and remained low during the summer. During September 2006, *Daphnia* spp. were observed in exceptionally high densities (197 ind. L^{-1}). A large early summer increase in *Daphnia* spp. density was observed in 2007, reaching 9 ind. L^{-1} during May 2007, after which *Daphnia* spp. were rarely present in the zooplankton community. *Ceriodaphnia* spp. (Figure 3.40c) were rare in Lake I2 except for summer 2007, when densities peaked at 32 ind. L^{-1} in August 2007. *Ceriodaphnia* spp. were also observed in autumn 2005 (6 ind. L^{-1}) and in low densities during 2006 (< 1 ind. L^{-1}). Calanoid copepods (Figure 3.40d) were present throughout most of the monitoring period, generally in densities less than 5 ind. L^{-1}. The highest densities of calanoid copepods were observed during the summers of July 2006 and August 2007, reaching 18 and 23 ind. L^{-1} respectively.

Cyclopoid copepods (Figure 3.40e) were more abundant in spring and late summer in all years. Peak abundances in 2005 and 2007 were 5 ind. L^{-1} and 4 ind. L^{-1} respectively, however in 2006 cyclopoid copepods were significantly more abundant, reaching 62 ind. L^{-1} in September 2006. The only members of the genus *Bosmina* (Figure 3.40f) observed in Lake I2 was *B. longirostris*. Occurrences of *B. longirostris* were mainly limited to late summer, although in 2007 *B. longirostris* was also recorded during early summer. The highest densities of *B. longirostris* occurred during the summer of 2006 (13 ind. L^{-1}), over twice that recorded during 2005 and substantially higher than during 2007. Rotifers (principally *Asplancha* spp.) were frequently observed in the zooplankton samples during 2006 and 2007 (Figure 3.40g). *Eurycercus lamellatus* and *Chydorus ovalis* were both present in the zooplankton community during June and July 2006, although at low densities. *Alona* spp. was recorded in October 2005.

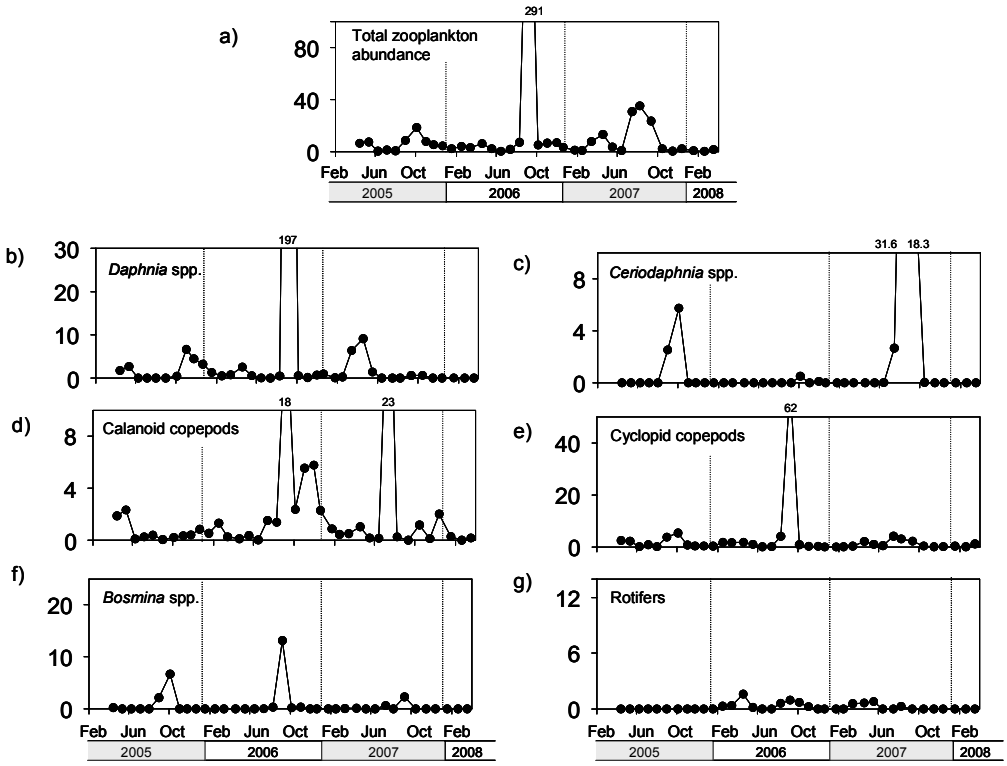


Figure 3.40 Timeseries of zooplankton abundance: a) total abundance; b-g) selected groups, March 2005-March 2008 in Lake I2 (Clifton Pond). All data are presented as individuals L^{-1}.

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3.7 Lake I3 (Beeston Pond)

Figure 3.41 A westerly view across Lake I3 (Beeston Pond). The culverted inflow site BPI is to the centre-left of the photograph. Lake I3 is isolated from the River Erewash.

3.7.1 Water chemistry

TP concentrations in Lake I3 (Figure 3.41) ranged between 54 and 746 µg L^{-1} during the monitoring period (mean, 228 ±22 µg L^{-1}, Figure 3.42). TP concentrations rose steadily from 108 µg L^{-1} in March 2005 to 326 µg L^{-1} by August 2005, then declined throughout winter and spring 2006 to 53 µg L^{-1} in April 2006. A sharp increase in TP concentrations occurred between April and May 2006. TP concentrations increased to a maximum of 746 µg L^{-1} in August 2007 then declined to 123 µg L^{-1} in January 2007. During 2007, TP increased and decreased between <100 µg L^{-1} (May 2007) and 450 µg L^{-1} (September 2007). TP concentrations decreased until March 2008. SRP concentrations (Figure 3.42a) were highly seasonal. The mean SRP concentration during the monitoring period was 112 (±13) µg L^{-1} and ranged between 3 and 346 µg L^{-1}. SRP concentrations rose from 4 µg L^{-1} in March 2005 to 210 µg L^{-1} in August, then decreased until spring 2006. SRP concentrations increased from <8 µg L^{-1} in March 2006 to 291 µg L^{-1} in May 2006. SRP concentrations reached a maximum of 346

µg L^{-1} in August 2006 then decreased sharply to 117 µg L^{-1} in September 2006. SRP concentrations decreased until spring 2007, then increased to 115 µg L^{-1} in July 2007. SRP concentrations reached a maximum of 238 µg L^{-1} in September 2007 and declined throughout the remainder of the monitoring period.

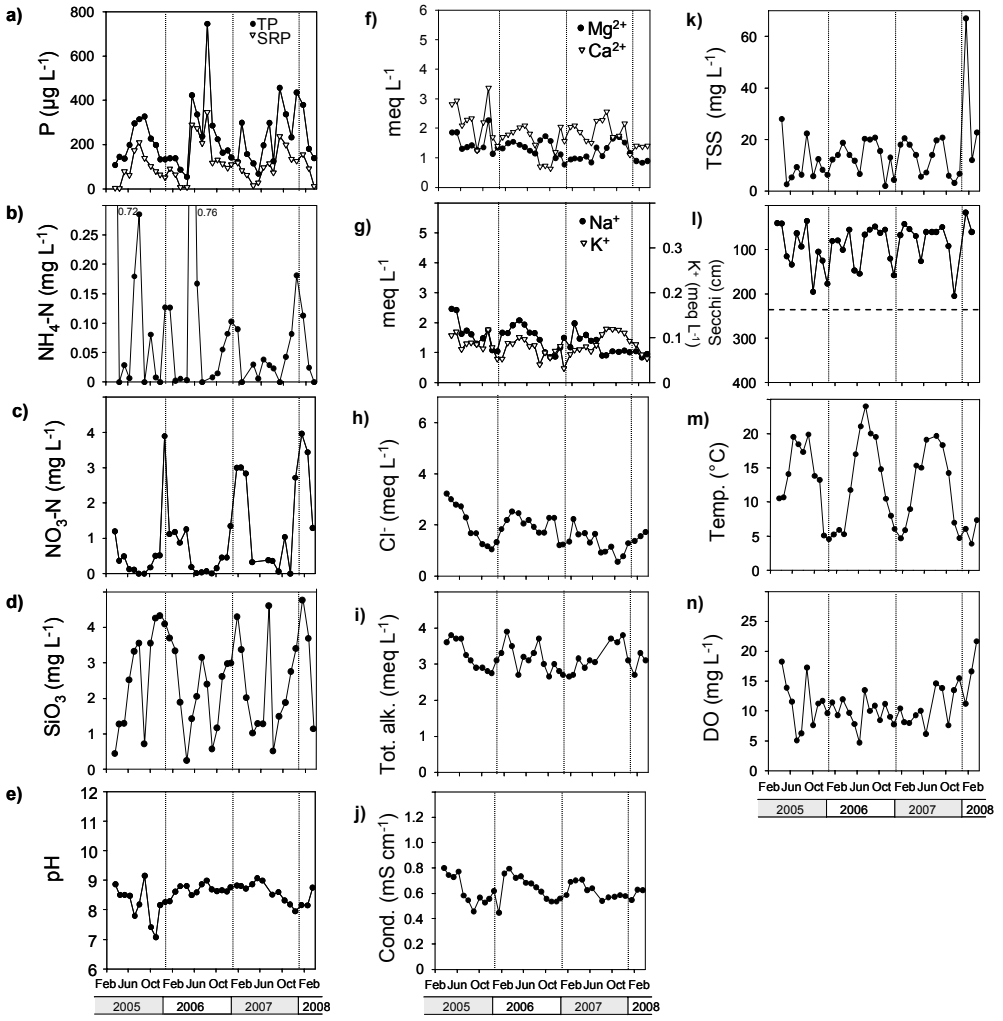

Figure 3.42 Water chemistry and physico-chemical properties of Lake I3 (Beeston Pond) March 2005 to March 2008: a) TP and SRP (µg L^{-1}); b) NH_4-N (mg L^{-1}); c) NO_3-N (mg L^{-1}); d) SiO_3 (mg L^{-1}); e) pH; f) Mg^{2+} and Ca^{2+} (meq L^{-1}); g) Na^+ and K^+ (meq L^{-1}); h) Cl^- (meq L^{-1}); i) total alkalinity (meq L^{-1}); j) specific conductivity (mS cm^{-1}); k) TSS (mg L^{-1}); l) Secchi disk depth (cm, dotted line represents approximate lake depth); m) surface temperature (°C); n) DO (mg L^{-1}).

NH_4-N concentrations (Figure 3.42b) were highly variable. The mean NH_4-N concentration during the monitoring period was 0.09 (±0.003) mg L^{-1}, ranging between 0 and 0.76 mg L^{-1}. Between March and April 2005 , NH_4-N concentrations fell from 0.72 mg L^{-1} to below detectable limits, and remained less than 0.03 mg L^{-1} until June 2005. From February to May 2006, NH_4-N concentrations were very low then rose sharply to reach 0.76 mg L^{-1} in May 2006 before decreasing rapidly. NH_4-N concentrations during 2007 were less variable than during 2005 and 2006, rising gradually between September 2006 and January 2007. NH_4-N concentrations did not exceed 0.05 mg L^{-1} until November 2007. NO_3-N concentrations (Figure 3.42c) ranged between 0 and 4.0 mg L^{-1} (mean 1.0 ±0.2 mg L^{-1}) during the monitoring period. The highest concentrations of NO_3-N were observed during the winter months. In 2005, NO_3-N concentrations fell from 1.2 mg L^{-1} in March to below detectable limits during August 2006. Then, increases in NO_3-N concentration were observed, with a substantial increase of 3.5 mg L^{-1} occurring between November and December. NO_3-N concentrations fell during spring and summer 2006. Similarly high winter concentrations and low summer concentrations occurred during 2007.

The mean SiO_3 concentration during the monitoring period was 2.4 (±0.2) mg L^{-1}, and ranged between 0.2 and 4.8 mg L^{-1} (Figure 3.42d). SiO_3 concentrations were highly seasonal. In each year, increases in SiO_3 concentrations were observed in early summer and winter and rapid decreases during the spring and autumn. SiO_3 concentrations rose from 0.4 mg L^{-1} in March 2005 to 3.5 mg L^{-1} in early August 2005. SiO_3 concentrations increased rapidly in November 2005. By April 2006, SiO_3 concentrations had decreased to 0.2 mg L^{-1}, although they rose to 3.1 mg L^{-1} in July 2006. SiO_3 concentrations were 0.5 mg L^{-1} in October 2006, after which SiO_3 increased to a maximum of 4.3 mg L^{-1} in January 2007. Increases in SiO_3 concentrations were followed by a sharp decrease to 0.5 mg L^{-1} in August 2007. SiO_3 concentrations rose to 4.8 mg L^{-1} in January 2008 before falling to 1.1 mg L^{-1} in March 2008.

pH (Figure 3.42e) in Lake I3 ranged between 7.1 and 9.15. The mean pH during the monitoring period was 8.51 ±0.07. Little seasonality was noted in pH. A fall in pH was observed from March 2005 to November 2005, except for a peak to 9.15 in August 2005. An increasing trend in pH was observed from winter 2005 to early summer

2007, after which pH fell progressively to 7.95 in December 2007, before rising to 8.75 by the end of the monitoring period. pH did not show any significant changes throughout the water column (Figure 3.43a).

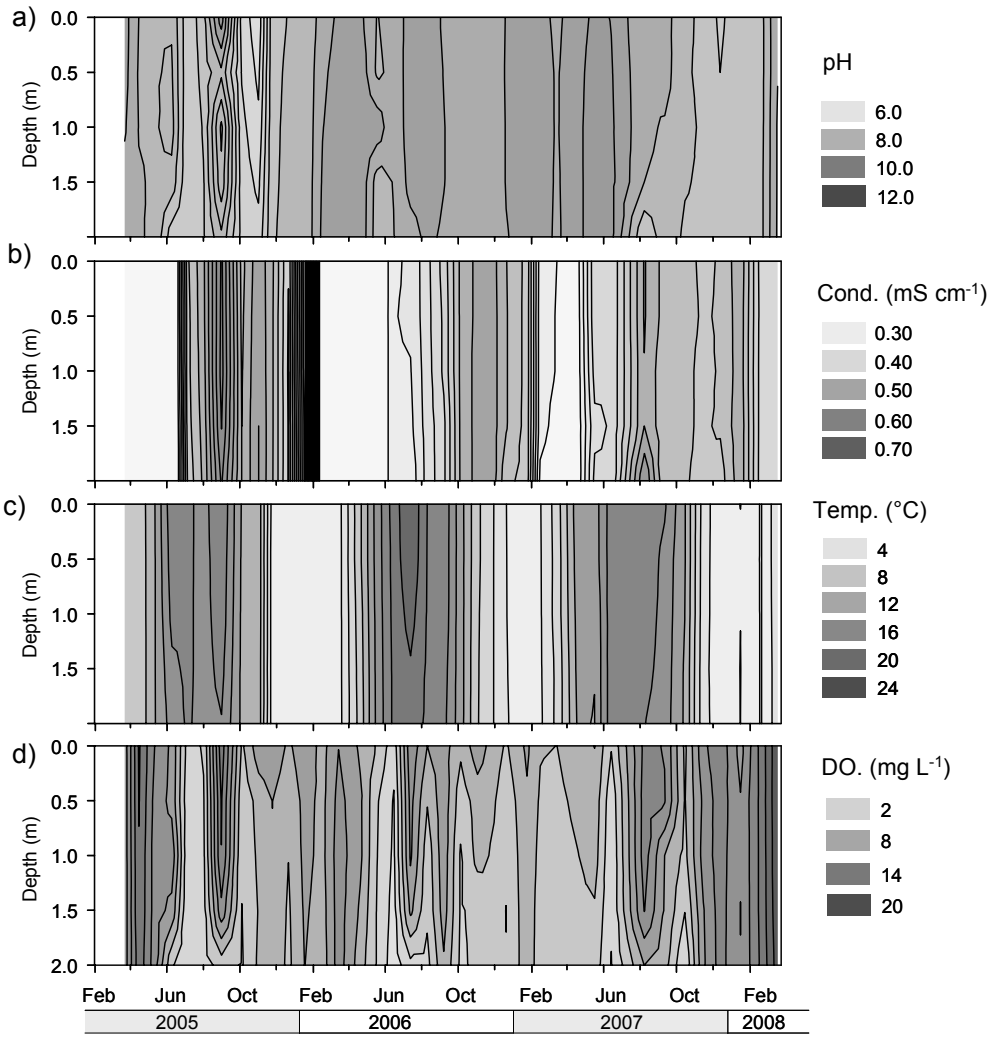

Figure 3.43 Profiles of a) pH; b) specific conductivity (mS cm^{-1}); c) temperature (°C); d) dissolved oxygen (mg L^{-1}) in Lake I3 (Beeston Pond) March 2005 to March 2008.

The mean Mg^{2+} concentration was 1.31 (±0.05) meq L^{-1} and ranged between 0.77 and 2.27 meq L^{-1} during the monitoring period (Figure 3.42f). A trend towards decreasing Mg^{2+} concentrations occurred (r_s = -0.456, p = 0.004). Peak concentrations of Mg^{2+} occurred during October and November each year and the lowest concentrations each year tended to occur in early summer. Ca^{2+} concentrations (Figure 3.42f) ranged between 0.63 and 3.37 meq L^{-1} during the monitoring period. The mean Ca^{2+}

concentration was 1.82 (±0.1) meq L^{-1}. An overall trend towards a decrease in Ca^{2+} concentrations was observed (r_s = -0.340, p = 0.037). The highest concentrations of Ca^{2+} were highest in April, May and November 2005, and the lowest (0.63–0.73 meq L^{-1}) during August to October 2006.

Na^+ concentrations (Figure 3.42g) declined throughout the monitoring period (r_s = -0.655, $p \leq 0.001$). The lowest concentrations (~1 meq L^{-1}) occurred in October and November during 2005 and 2006, and July 2007, whilst maximum concentrations (in excess of 2 meq L^{-1}) tended to occur in March and April each year. In contrast to 2005 and 2006, Na^+ concentrations did not increase during winter 2007 but remained relatively constant. K^+ concentrations (Figure 3.42g) showed a decreasing trend from March 2005 until the beginning of 2007, after which a steep increase in K^+ concentrations occurred until August 2007 (0.12 meq L^{-1}) before a progressive decline to 0.05 meq L^{-1} in March 2008. During 2005 and 2006, a seasonal trend of higher K^+ concentrations in early summer and early autumn was observed. The mean K^+ concentration throughout the monitoring period was 0.82 (±0.04) meq $L^{-1,}$ and ranged between 0.03 and 0.12 meq L^{-1}. Cl^- concentrations (Figure 3.42h) showed a declining trend throughout the monitoring period (r_s = -0.511, p = 0.001). The mean Cl^- concentration was 1.7 (±0.1) meq L^{-1} and ranged between 0.6 and 3.0 meq L^{-1}. Cl^- concentration decreased steadily from March 2005 to November 2005 from 3.2 to 1.0 meq L^{-1}, then rose to 2.5 meq L^{-1} in March 2006. Cl^- concentrations decreased erratically throughout 2006 and 2007, then increased from 0.6 meq L^{-1} in October 2007 to 1.7 meq L^{-1} in March 2008. Total alkalinity ranged between 2.65 and 3.90 meq L^{-1} (mean, 3.17 ±0.06 meq L^{-1}) during the monitoring period. A general decline in total alkalinity occurred from the beginning of the monitoring period until December 2005, after which alkalinity became more variable. Maximum total alkalinity was observed during February and November 2006 (3.9 and 3.7 meq L^{-1} respectively). Total alkalinity gradually increased throughout 2007, before declining sharply from 3.8 meq L^{-1} in November 2007 to 2.7 meq L^{-1} in January 2008.

Specific conductivity (Figure 3.42j) ranged between 0.45 and 0.80 mS cm^{-1} during the monitoring period. The mean specific conductivity was 0.62 (±0.01) mS cm^{-1}. Specific conductivity tended to be highest in February and March each year, and then declined throughout the summer before rising rapidly from mid to late winter. In 2007, peak

specific conductivity (0.71 mS cm^{-1}) was less than in 2005 and 2006 (0.80 and 0.79 mS cm^{-1} respectively). Specific conductivity remained largely homogenous throughout the water column (Figure 3.43b).

TSS concentrations (Figure 3.42k) ranged between 2 and 67 mg L^{-1} (mean, 13.7 ±1.8 mg L^{-1}), although until December 2007 did not exceed 28 mg L^{-1}. Higher TSS concentrations occurred in late winter and early spring, and mid summer, typically reaching approximately 20 mg L^{-1}. An abrupt increase in TSS concentrations occurred between December 2007 (7 mg L^{-1}) and January 2008 (67 mg L^{-1}) followed by a decline to 12 mg L^{-1} in February 2008.

The mean Secchi disk depth in Lake I3 was 88 (±8) cm and ranged between 16 and 205 cm (Figure 3.42l). Secchi disk depths were characterised by increases during early summer and late autumn to winter. During 2005, Secchi disk depths increased from 40 cm in March to 134 cm in June. Secchi disk depths decreased to 35cm in August 2005 then increased rapidly to 195 cm in October 2005. Secchi disk depths decreased over the autumn and winter, increased to ~ 150 cm during May and June, then decreased again to 48–66 cm between July and November 2006. Secchi disk depths increased to 158 cm in December 2006, and reached 126 cm in May 2007. Secchi depths increased sharply to 205 cm in December 2007 and then fell rapidly to 16 cm in January 2008.

The mean surface water temperature (Figure 3.42m) in Lake I3 was 12.3 (±1.0) °C, and ranged between 3.9 and 24.0 °C during the monitoring period. Temperatures peaked in each year during July to August, although the highest temperature during 2006 was around 4°C higher than in 2005 and 2007. Minimum temperatures occurred during November to February each winter, and were generally ~5 °C. Water temperature was largely homogenous throughout the water column during the monitoring period (Figure 3.43c). During summer 2005 and 2006 temperature gradients of approximately 2 °C were observed between the surface at 2 m depth, although during summer 2007 temperature gradients were insignificant.

DO concentrations (Figure 3.42n) ranged between 4.7 and 13.5 mg L^{-1} (mean, 12.3 ±1.0 mg L^{-1}). DO concentrations were variable throughout 2005 and 2006. From

October 2007 to the end of the monitoring period DO concentrations gradually increased, reaching 21.7 mg L^{-1} during March 2008. DO concentrations were generally greater on the water surface than near the bottom of the lake (Figure 3.43d). Gradients in DO concentrations were strongest during the summer of 2005 and 2006 when differences of approximately 9 mg L^{-1} were observed between the water surface and at 2 m depth.

3.7.2 Phytoplankton

Chlorophyll-*a* concentration

Chlorophyll-*a* concentrations (Figure 3.44) were highly variable throughout the monitoring period. During 2006 and 2007, chlorophyll-*a* concentrations generally reached maxima in early summer and late summer to autumn. In 2005, a seasonal trend was not so clear. During 2005, the highest chlorophyll-*a* concentrations occurred during April (70 µg L^{-1}), then decreased to 16 µg L^{-1} in October 2005. Chlorophyll-*a* concentrations rose by 49 µg L^{-1} between October and November 2005, then decreased to less than 5 µg L^{-1} Chlorophyll-*a* concentrations reached a maximum between February and April 2006 and again during September 2006.

A rapid decline in chlorophyll-*a* concentrations occurred between September and November 2006 to 9 µg L^{-1}. Concentrations of chlorophyll-*a* increased throughout summer 2007, reaching a maximum of 85 µg L^{-1} in August 2007, and then decreased to 7 µg L^{-1} in December. Between January and March 2008, chlorophyll-*a* concentrations rose from 8 to 78 µg L^{-1}. The mean chlorophyll-*a* concentrations during the monitoring period was 38.9 (±4.3) µg L^{-1}.

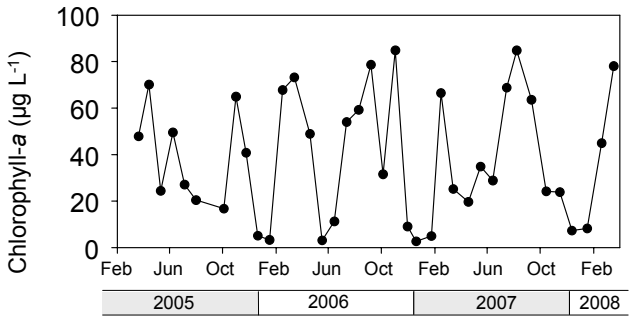

Figure 3.44 Chlorophyll-*a* concentrations in Lake I3, March 2005 to March 2008.

Community composition

The phytoplankton community in Lake I3 was dominated by diatoms (Figure 3.45). Diatoms were generally most abundant during spring and late summer. Cryptophytes were also abundant in the phytoplankton, particularly during the summer of 2007. Chlorophytes were not abundant in Lake I3 although were frequently observed in the phytoplankton community during spring and summer. Cyanobacteria contributed little to the phytoplankton community throughout the monitoring period. Dinophytes and chrysophytes were both frequently found in the phytoplankton, particularly during spring.

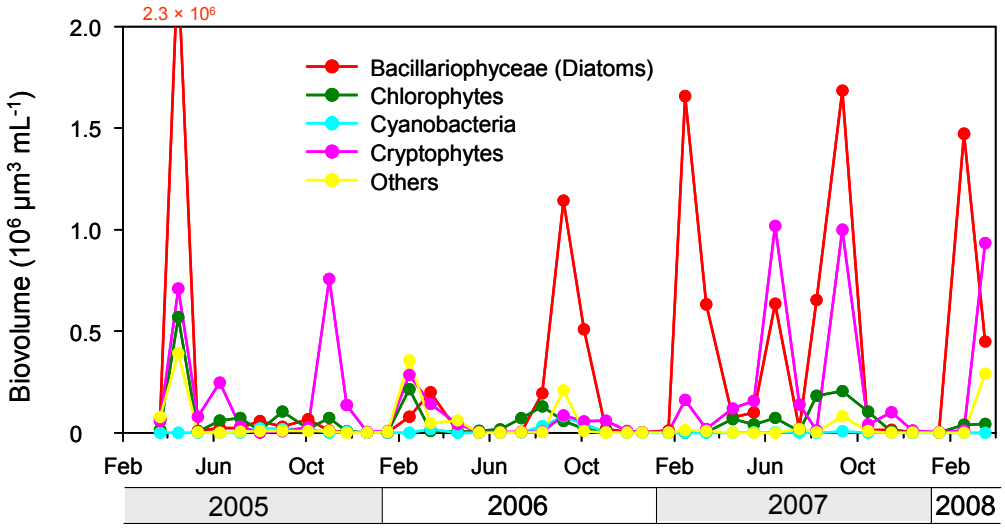

Figure 3.45 Summary of total phytoplankton group biovolumes in Lake I3, March 2003 to March 2008. 'Bacillariophyceae' refers to all diatoms, and 'others' comprises dinophytes and chrysophytes.

Diatoms (Figure 3.46a-e) became increasingly abundant throughout the monitoring period. Centric diatoms (Figure 3.46a) were present throughout, although they were relatively scarce during 2005. However, a significant biovolume of centric diatoms was present during April 2005 (7.7×10^5 µm^3 mL^{-1}). A slight increase in the abundance of centric species was observed during late summer and autumn 2005 (peaking at 3×10^4 µm^3 mL^{-1} in August). Centric diatoms were scarce during winter 2005 and briefly increased in biovolume to 1.9×10^5 µm^3 mL^{-1} in April 2006. In early summer, centric species were virtually absent and then increased to 4.4×10^5 µm^3 mL^{-1} in September 2006. A large spring peak in centric diatom biovolume occurred in February 2007 (1.4×10^6 µm^3 mL^{-1}). Although rare during early summer 2007, centric diatoms again

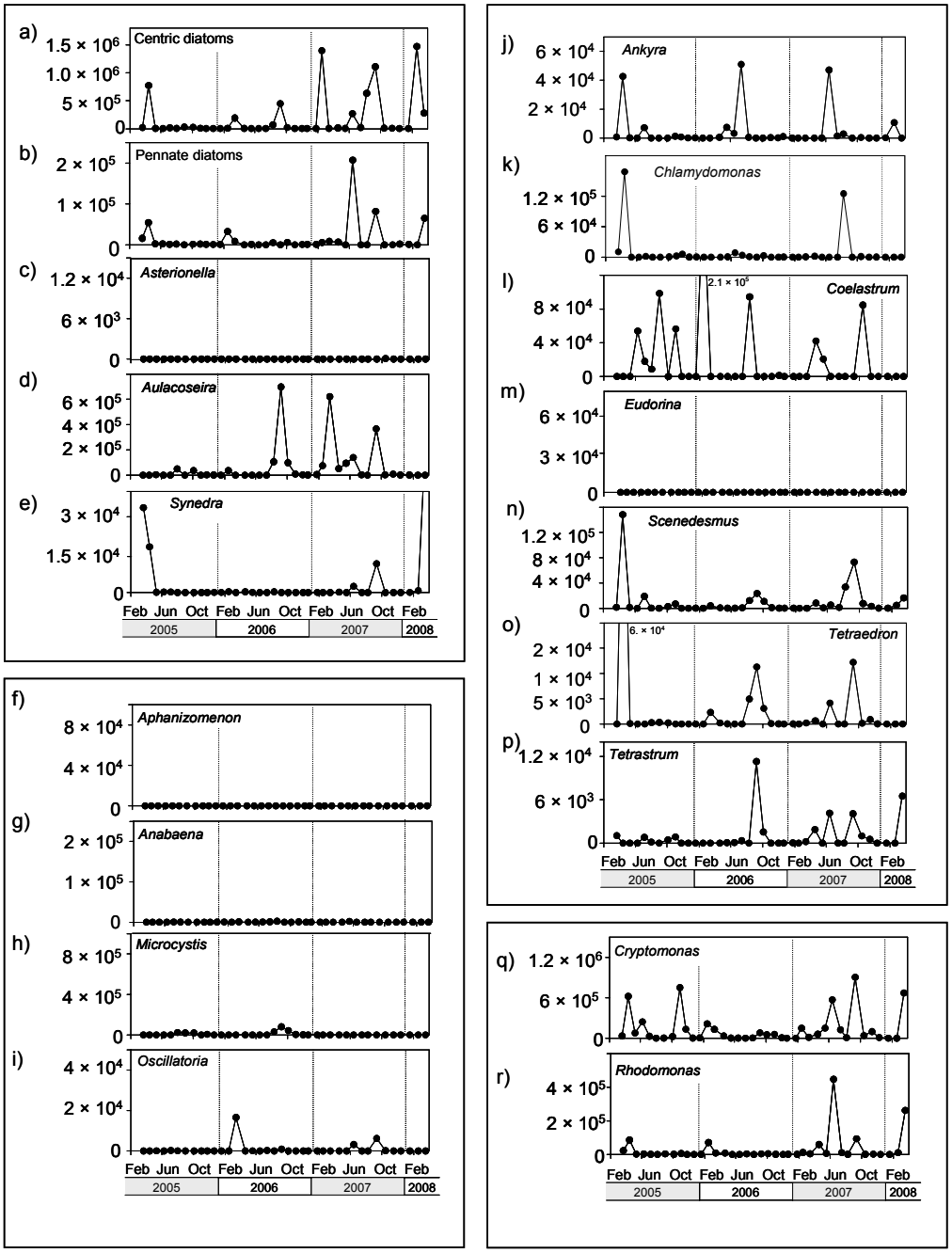

Figure 3.46 Time series of biovolumes (µm^3 mL^{-1}) of selected phytoplankton species in Lake I3 (Beeston Pond), March 2005-March 2008: a-e) Bacillariophyceae (diatoms); f-i) cyanobacteria; j-p) Chlorophyceae; q-r) Cryptophyceae.

became abundant during August and September 2007. A large increase in biovolume occurred in February 2008 to over 1.4 × 10^6 µm^3 mL^{-1}. Pennate diatoms (Figure 3.46b) were similarly rare during 2005 and 2006 and were generally less abundant than

centric species. Isolated peaks in abundance occurred in April 2005 and February 2006. Pennate species were observed in substantial quantities during June 2007 and again in September 2007. *Asterionella* sp. (Figure 3.46c) was not observed in significant biovolumes during the monitoring period. *Aulacoseira* spp. (Figure 3.46d) became more abundant during 2006 reaching a maximum of 7.0×10^6 μm^3 mL^{-1}. *Synedra* spp. (Figure 3.46e) were present in significant biovolumes during March and April 2005, and were virtually absent until a brief increase in biovolume during September 2007.

Cyanobacteria were rare throughout the monitoring period in Lake I3. *Aphanizomenon* spp. (Figure 3.46f) were absent from the phytoplankton community. *Anabaena* sp. (Figure 3.46g) was not observed in significant biovolumes, although was present at biovolumes less than 1000 μm^3 mL^{-1} during the summer of 2006. *Microcystis* sp. (Figure 3.46h) was the most abundant cyanobacterium in Lake I3, present during late summer 2005 and summer 2006 at biovolumes less than 4.3×10^4 μm^3 mL^{-1}. *Microcystis* sp. was not present during 2007. *Oscillatoria* sp. (Figure 3.46i) was observed only in small biovolumes during the monitoring period. The highest biovolume of *Oscillatoria* sp. occurred during March 2006 (1.7×10^4 μm^3 mL^{-1}). During 2007, *Oscillatoria* sp. was present during summer and autumn, peaking at 6.1×10^3 μm^3 mL^{-1} in September 2007.

Chlorophytes (Figure 3.46j-p) were a significant component of the phytoplankton community for much of the monitoring period. *A. judyaii* (Figure 3.46j) was frequently observed. The highest biovolumes of *A. judyaii* tended to occur during early summer, although in 2005, the highest biovolume of the species occurred during March. In each year of the monitoring period, *A. judyaii* was present in similar biovolumes ($4.3\text{-}5.1 \times 10^4$ μm^3 mL^{-1}). A brief increase in the biovolume of *A. judyaii* occurred during February 2008. *Chlamydomonas* spp. (Figure 3.46k) were present throughout most of the monitoring period. The highest biovolumes of *Chlamydomonas* spp. were recorded during April 2005 (1.7×10^5 μm^3 mL^{-1}) and August 2007 (1.5×10^5 μm^3 mL^{-1}). *Chlamydomonas* spp. were less abundant during 2006, peaking at only 8×10^3 μm mL^{-1}. *Coelastrum* sp. (Figure 3.46l) was frequently an abundant member of the phytoplankton community. During 2005, *Coelastrum* sp. was present in the phytoplankton from June to November. *Coelastrum* sp. was also abundant during

February 2006 (2.1×10^5 µm^3 mL^{-1}) and August 2006 (9.4×10^4 µm^3 mL^{-1}). In 2007, *Coelastrum* sp. was observed during early and late summer, reaching a maximum of 8.6×10^5 µm^3 mL^{-1} in October. *Eudorina* sp. (Figure 3.46m) was not observed in Lake I3 during the monitoring period. *Scenedesmus* spp. were frequently encountered in the phytoplankton community. The genera was mainly represented by the species *S. opoliensis* and *S. communis*. The highest biovolume of *Scenedesmus* spp. occurred during March 2005. During August and September 2006, *S. opoliensis* was dominant, and total biovolume of *Scenedesmus* spp. reached 2.3×10^4 µm^3 mL^{-1} in September 2006. *Scenedesmus* spp. increased in biovolume during April 2007 and during late summer 2007. The genus *Tetraedron* (Figure 3.46o) was dominated by *T. triangulare*, although *T. caudatum*, *T. regulare* and *T. minimum* were also observed. *Tetraedron* spp. were abundant during spring 2005 and were rarely encountered until spring 2006. *Tetraedron* spp. became abundant between August and October. A similar sequence of abundance of *Tetraedron* spp. was observed during 2007. *Tetrastrum* sp. (Figure 3.46p) was observed during 2005 and 2006 although in small biovolumes, and in March 2008 the biovolume had increased to 6.5×10^3 µm^3 mL^{-1}. *Pediastrum* spp. were often present in the phytoplankton community during summer months of 2005 and 2007 and contributed substantially to the total biovolume of chlorophytes. *Monoraphidium* sp. was present through most of the monitoring period, although in small biovolumes. Unidentified flagellates were frequently observed in the phytoplankton of Lake I3, and were most abundant during April 2005 (1.2×10^6 µm^3 mL^{-1}) and throughout 2007 (maximum biovolume of 1.5×10^4 µm^3 mL^{-1} in June). Small green flagellates were also abundant during March 2008.

Cryptophytes (Figure 3.46q-r) were present throughout the monitoring period and were comparatively a highly abundant group in the phytoplankton community. *Cryptomonas* spp. (Figure 3.46q) increased in biovolume between March and April 2005, to reach 6.2×10^5 µm^3 mL^{-1}, then fell throughout the summer until September 2005 after which a sharp increase in biovolume was observed to 7.5×10^5 µm^3 mL^{-1} in November 2005. During 2006, *Cryptomonas* spp. were present in smaller biovolumes than 2005. During 2007 *Cryptomonas* spp. were abundant in June and September, and in March 2008 a sharp increase in *Cryptomonas* spp. biovolume occurred. *Rhodomonas* spp. (Figure 3.46r) were present throughout the monitoring period although in smaller biovolumes than *Cryptomonas* spp. During 2005 and 2006,

Rhodomonas spp. were most abundant during April and February respectively. During 2007, a similar peak in *Rhodomonas* spp. biovolume occurred in April 2007 and were then highly abundant during June 2007. Between August and September 2007 the biovolume of *Rhodomonas* spp. rose but decreased during the winter of 2007. Increases were observed during early 2008.

Euglena spp. were present in the phytoplankton community in 2005 and 2006, although rarely contributed significantly to the total phytoplankton biovolume. *Phacus* spp. were occasionally observed throughout the monitoring period in small biovolumes, generally less than 1.5×10^3 μm^3 mL^{-1}. In February 2006, over 3.3×10^5 μm^3 mL^{-1} of *P. caudatum* was observed. *Phacus* spp. were also abundant in March 2008, reaching 7.1×10^3 μm^3 mL^{-1}. *Trachelomonas* sp. was also relatively common in the phytoplankton of Lake I3 in spring, particularly during March 2005 (1.4×10^4 μm^3 mL^{-1}), 2006 (1.8×10^4 μm^3 mL^{-1}) and 2008 (1.7×10^5 μm^3 mL^{-1}). The dinophyte community was represented by the genera *Gymnodinium* and *Peridinium*. Both genera were abundant during the spring of 2005. *Peridinium* spp. was generally the most abundant dinoflagellate, particularly during March 2006 (2.9×10^4 μm^3 mL^{-1}) and September 2006 (1.9×10^5 μm^3 mL^{-1}). The chrysophyte *Mallomonas* sp. was most abundant during April 2005 (2.7×10^5 μm^3 mL^{-1}) and in September 2007 (4.9×10^5 μm^3 mL^{-1}).

3.7.3 Macrophytes

Submerged macrophytes were rare in Lake I3. Three submerged species (*Callitriche stagnalis*, *Fontinalis antipyretica* and *Potamogeton trichoides*) were found. The floating-leaved community entirely comprised *P. amphibium*, which was only observed in the northernmost areas of the lake. Emergent macrophytes were found along most of the shores. *Acorus calamus* was found mainly along southern shores of the lake. *Sparganium erectum* and *Typha* spp. (*T. angustifolia* and *T. latifolia*) were found throughout the lakes shores. *Glyceria maxima* was found at three sampling sites, and *Carex* sp. and *Juncus* sp. were found mainly along the northern and eastern shores of the lake. *Iris pseudcorus* and *Menthe aquatica* were each recorded once in the north-east of Lake I3.

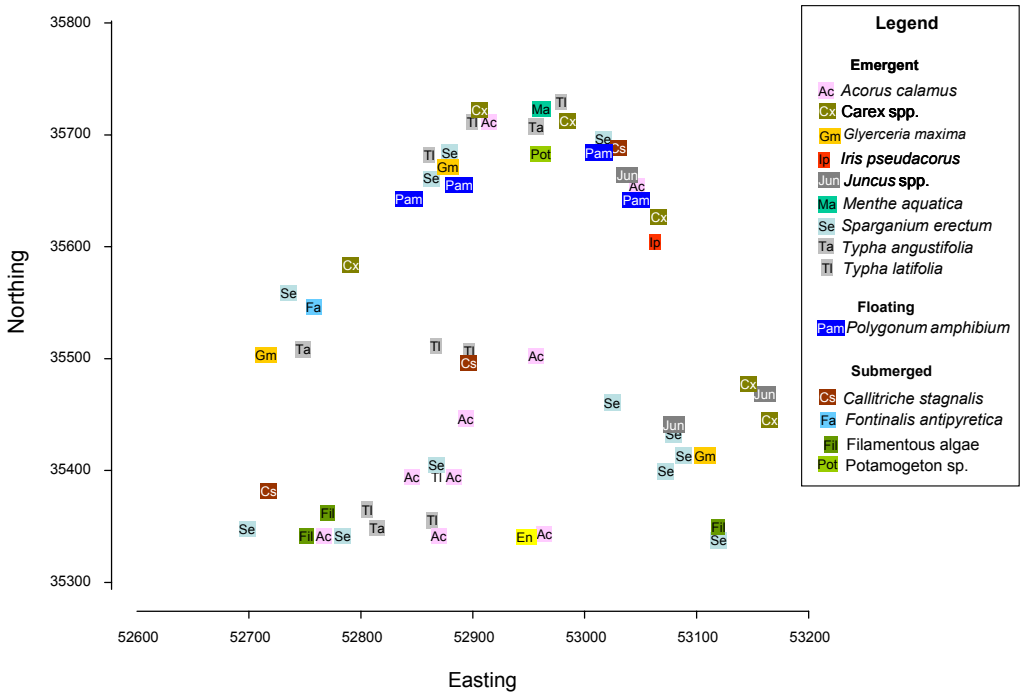

Figure 3.47 Schematic map of the occurrence of aquatic macrophytes observed in Lake I3 (Beeston Pond).

3.7.4 Zooplankton

Total zooplankton abundance in Lake I3 (Figure 3.48a) ranged between 0 and 49 ind. L^{-1} during the monitoring period. During 2005, zooplankton were most abundant during April 2005 and lowest during summer. After increasing to 17 ind. L^{-1} during October 2005, total zooplankton biomass declined erratically throughout the winter of 2005. Total zooplankton abundance then increased rapidly to 40 ind. L^{-1} in May 2005, and remained low until spring 2007. An increase in total zooplankton abundance was observed between March and April 2007, reaching 49 ind. L^{-1} in April 2007. Total zooplankton abundance fell to between 5 and 17 ind. L^{-1} until September, and decreased throughout autumn and winter 2007.

Daphnia spp. in Lake I3 (Figure 3.48b) comprised *D. hyalina* and *D. cucullata*. *D. hyalina* was generally the more abundant. The abundance of *Daphnia* spp. fell from 3.6 ind. L^{-1} in March 2005 to less than 0.4 ind. L^{-1} throughout the summer of 2005, then increased to 7 ind. L^{-1} during November and December 2005. 24 ind. L^{-1} were observed in June 2006, comprising *D. cucullata.* Similarly to 2005 and 2006, *Daphnia*

spp. densities increased to peak at 8 ind. L^{-1} (mainly *D. cucullata*) in November 2007 before falling throughout the rest of the monitoring period. *Ceriodaphnia* spp. (Figure 3.48c) were comparatively rare throughout the monitoring period compared to other zooplankton groups. During 2005, *Ceriodaphnia* spp. were rarely found in the zooplankton community. *Ceriodpahnia* spp. were observed in August 2006 after which *Ceriodaphnia* spp. were not recorded until February 2007. During July to September 2007, *Ceriodaphnia* spp. were occasionally present.

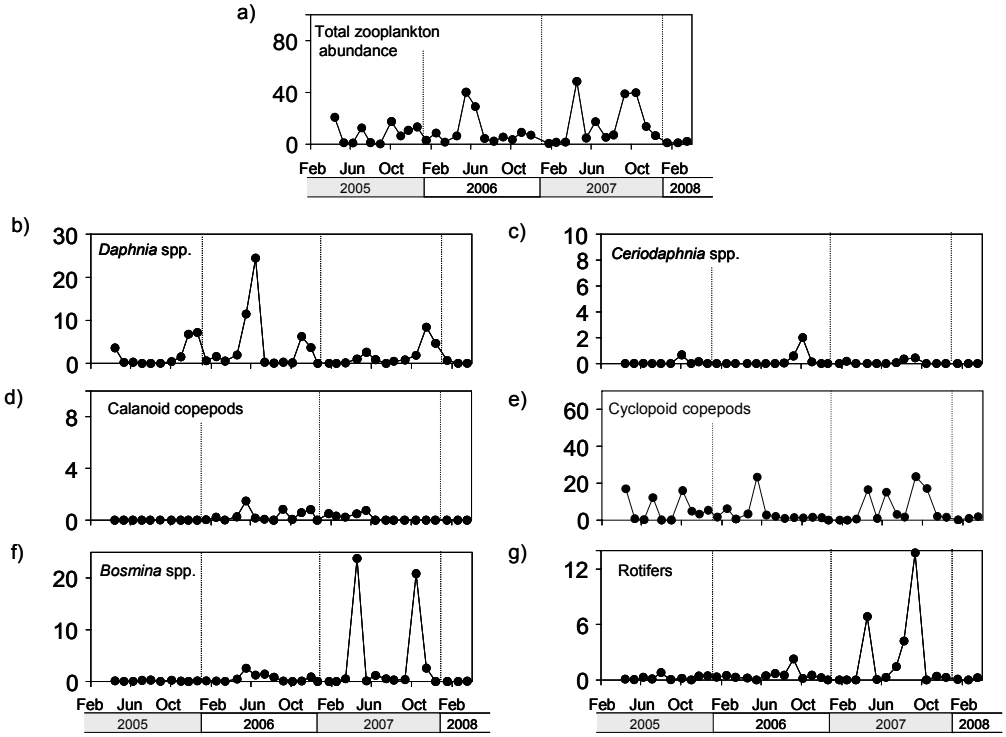

Figure 3.48 Timeseries of zooplankton abundance: a) total abundance; b-g) selected groups, March 2005-March 2008 in Lake I3 (Beeston Pond). All data are presented as individuals L^{-1}.

Calanoid copepods (Figure 3.48d) were only observed in significant densities between February 2006 and May 2007. The highest density of calanoid copepods was recorded in May 2006 (1.5 ind. L^{-1}). The abundance of cyclopoid copepods (Figure 3.48e) was highly variable. During 2005, the abundance of cyclopoid copepods fell from 17 ind. L^{-1} during April to <1 ind. L^{-1} during May, June and August. A brief increase to 12 ind. L^{-1} occurred during July 2005. From September 2005 to March 2006, cyclopoid copepod density fell from 16 to <1 ind. L^{-1}. An abrupt increase in the

density of cyclopoid copepods occurred between April and May 2006, peaking at 23 ind. L^{-1}, after which a decline to 3 ind. L^{-1} was observed, followed by a progressive decline throughout winter 2006 to <1 ind. L^{-1} in March 2007. Peaks in the density of cyclopoid copepods to over 15 ind. L^{-1} occurred in April, June and September 2007, followed by rapid decreases in abundance. From November 2007 until March 2008 the density of cyclopoid copepods did not exceed 2 ind. L^{-1}. The *Bosmina* genus (Figure 3.48f) almost exclusively comprised *B. longirostris* except for an isolated occurrence of *B. coregoni* in April 2006. *Bosmina* spp. were not observed during 2005 at densities more than 0.3 ind. L^{-1}. A brief increase in the density of *Bosmina* spp. to 3 ind. L^{-1} was observed during May 2006, then decreased gradually until the genera was absent in January and February 2007. A sharp increase in the abundance of *B. longirostris* was observed in April 2007 (to 24 ind. L^{-1}). Between May and September 2007, *B. longirositris* densities remained fewer than 2 ind. L^{-1}, then increased to 21 ind. L^{-1} between September and October 2007. From December 2007 to March 2008 *B. longirostris* was not observed in the zooplankton community at densities greater than 0.06 ind. L^{-1}.

Rotifers (Figure 3.48g) were rare from March 2005 to August 2006, never exceeding 1 ind. L^{-1}. In September 2006 2.2 ind. L^{-1} were found in the zooplankton sample. From October 2006 to March 2008 no more than 0.5 ind. L^{-1} were found, with no rotifers observed from January to March 2007. Rotifers became briefly more abundant in April 2007 (reaching 6.8 ind. L^{-1}) before increasing from <0.1 to 13.8 ind. L^{-1} between May and September 2007. From October 2007 to March 2008, rotifer densities did not exceed 0.4 ind. L^{-1}.

3.8 Comparisons of phytoplankton community structure among lakes

In order to compare the phytoplankton community structure between lakes correspondence analysis was used. Prior to performing CA, a detrended correspondence analysis (DCA) was used to assess the length of the first axis. Gradients that are greater than 2 standard deviations suggest a unimodal species response and datasets are therefore considered appropriate for unimodal analysis

techniques (Jongman et al., 1987). DCA showed the first axis to be 4.203 standard deviations long. CA has been used in other studies of phytoplankton communities (e.g. Fukishima et al., 1999; Takamura et al., 2003) and allows for associations between phytoplankton species and samples to be identified. Individual plots of phytoplankton species and samples for each lake were produced to facilitate this. In this analysis, axes 1 and 2 identified by CA explained 6.1% and 5.3% of correlation between sample and species scores, and had eigenvalues of 0.458 and 0.395 respectively. The first four axes cumulatively explained 21.4% of sample and species scores correlation, and total inertia was 7.485.

Figure 3.49 shows that a number of phytoplankton genera observed during the monitoring period formed a single cluster after CA. This included species that were typically observed during summertime, particularly in connected lakes. Chlorophyte genera, such as *Scenedesmus*, *Coelastrum* and *Pediastrum*, formed a cluster towards the left of the plot. Single-celled chlorophytes (including *Ankyra*, *Monoraphidium* and green flagellates) generally scored more highly on axis 1. The genera *Eudorina* and *Schroderia* were the only chlorophytes to be strongly associated with high axis 1 scores. *Closterium*, *Oocystis* and *Dictyosphaerium* all had negative axis 1 and two scores, and *Tetrastrum* had a positive axis 2 and negative axis 1 score.

Negative axis 1 scores were associated with centric diatoms, *Tabellaria*, *Asterionella* and *Aulacoseira*, whilst pennate diatoms, *Synedra*, *Cocconeis* and *Meridion* had positive axis 1 scores. Axis 1 separated cyanobacteria genera into *Microcystis* and *Oscillatoria*, both of which scored negatively on axis 1, and *Anabaena* and *Aphanizomenon* with positive axis 1 scores. The genera *Cryptomonas* and *Rhodomonas* both scored positively on axis 1 and were closely associated with chlorophytes and centric diatoms. *Peridinium* and *Phacus* were found to be closely associated with large chlorophytes and centric diatoms. *Trachelomonas*, *Euglena* and *Dinobryon* all had high axis 1 scores. Axis 2 separated the genera *Fragilaria*, *Monoraphidium*, *Gymnodinium*, *Dinobryon* and green flagellates from other genera.

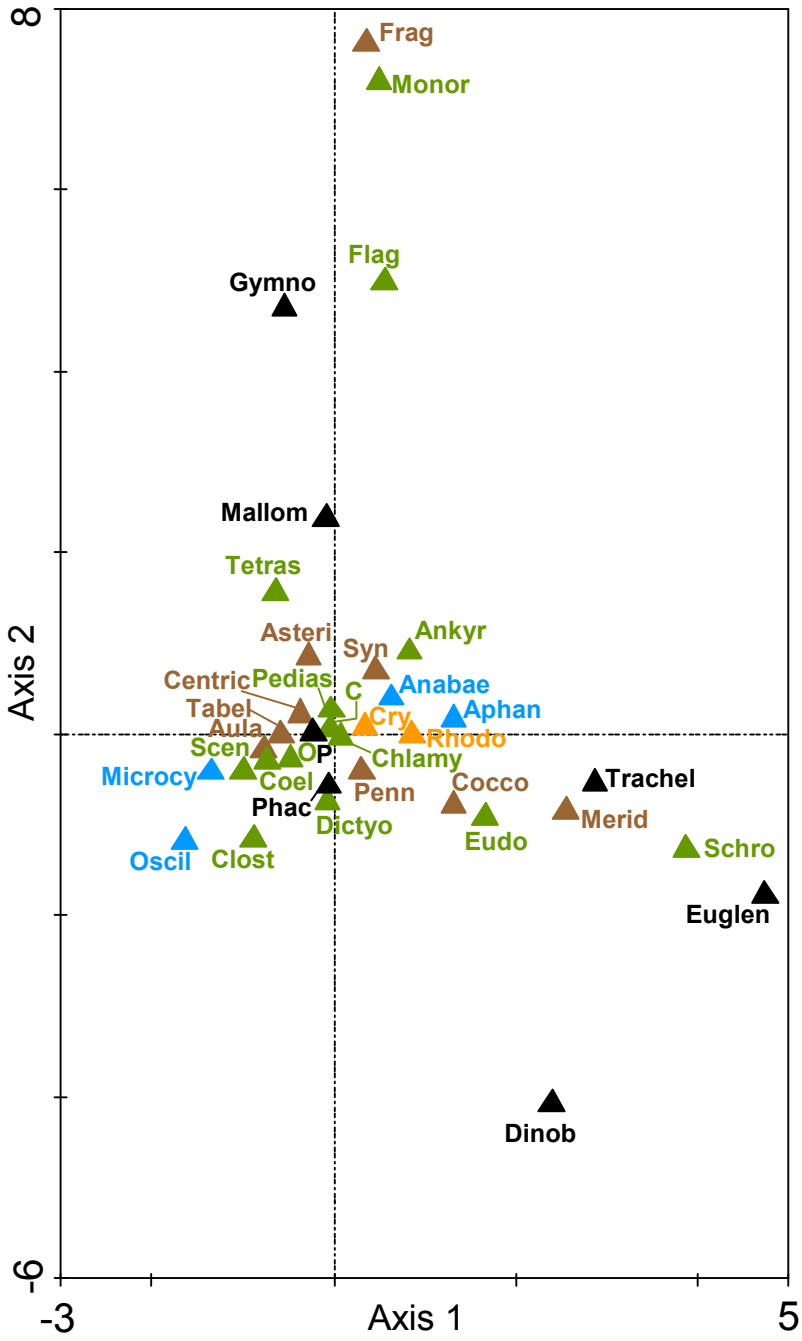

Figure 3.49 CA plot of phytoplankton genera from all lakes, March 2005 to March 2008, scaled by inter-sample distance using Hill's scaling. Phytoplankton genera are colour coded by group: brown, Bacillariophyceae; green, chlorophytes; orange, cryptophytes; blue, cyanobacteria; black, others (including chrysophytes and dinoflagellates). Abbreviations are given in Table 3.1.

Table 3.1 Phytoplankton genera abbreviations used in Figure 3.49.

Abbreviation	Genus
Bacillariophyceae	
Asteri	*Asterionella*
Aula	*Aulacoseira*
Centric	Centric diatoms
Cocco	*Cocconeis*
Frag	*Fragilaria*
Merid	*Meridion*
Penn	Pennate diatoms
Syn	*Synedra*
Tabel	*Tabellaria*
Chlorophyceae	
Ankyr	*Ankyra*
C	*Chlorella*
Chlamy	*Chlamydomonas*
Clost	*Closterium*
Coel	*Coelastrum*
Dictyo	*Dictyosphaerium*
Eudo	*Eudorina*
Flag	Unidentified flagellates
Monor	*Monoraphidium*
O	*Oocystis*
Pedias	*Pediastrum*
Scen	*Scenedesmus*
Schro	*Schroderia*
Tetras	*Tetrastrum*
Cyanobacteria	
Anabae	*Anabaena*
Aphan	*Aphanizomenon*
Microcy	*Microcystis*
Oscil	*Oscillatoria*
Cryptophytes	
Cry	*Cryptomonas*
Rhodo	*Rhodomonas*
Others	
Dinob	*Dinobryon*
Euglen	*Euglena*
Gymn	*Gymnodinium*
Mallom	*Mallomonas*
P	*Peridinium*
Phac	*Phacus*
Trachel	*Trachelomonas*

Sample scores for all connected lakes were generally negative on both axes (Figure 3.50a-c). Some seasonal differences were observed in sample scores, although separate clusters did not occur. Spring samples were generally located near the centre of the CA plots implying that the phytoplankton community was associated with diatoms, chlorophytes and cryptophytes. In Lake C1, samples taken during February and March 2008 were both distinct from other samples, reflecting the dominance of *Asterionella*, *Synedra*, and centric species of diatom. The March 2008 sample from Lake C3 contained a substantial quantity of unidentified green flagellates which explains the high axis 2 score and different position from March samples during other years. During May and June, sample scores were higher on axis 1, reflecting the importance of *Ankyra* and *Synedra*. In all connected lakes the lowest scores on axis 1 were associated with late summer samples, reflecting the dominance of chlorophytes, including genera such as *Scenedesmus* and *Coelastrum* and diatoms. August 2006 was an outlier from other summer samples in all connected lakes, most likely due to the occurrence of a high biovolume of *Oscillatoria* observed during late summer 2006. In lakes C2 and C3 (Figure 3.50b and c), the August 2007 samples, which had higher axis 1 scores than other summertime samples, reflected the comparatively high biomass of *Tetrastrum*, *Aulacoseira* and other centric diatoms. This was not evident in Lake C1 were the August 2008 sample was more closely associated with summer samples from other years. Winter samples were generally located near the centre of the cluster of samples reflecting the persistence of diatoms throughout the winter months. In both lakes C2 and C3, the June 2005 sample was an outlier from other early summer samples. In both lakes, unidentified green flagellates were abundant which explains their relatively high axis 2 scores.

Seasonal patterns were less evident in isolated lakes I1 and I2 (Figure 3.51a and b). CA scores were generally higher on axis 1 than in connected lakes, although axis 2 scores were not substantially different. In Lake I2, summertime scores on axis 1 were low, particularly during 2006 when the cyanobacteria *Oscillatoria* was abundant. During other years and in Lake I1, summertime scores were not as low, which reflects the higher biovolumes of chlorophytes observed. Spring and autumn samples were not clearly distinguished from other samples in either Lake I1 or Lake I2. In Lake I1,

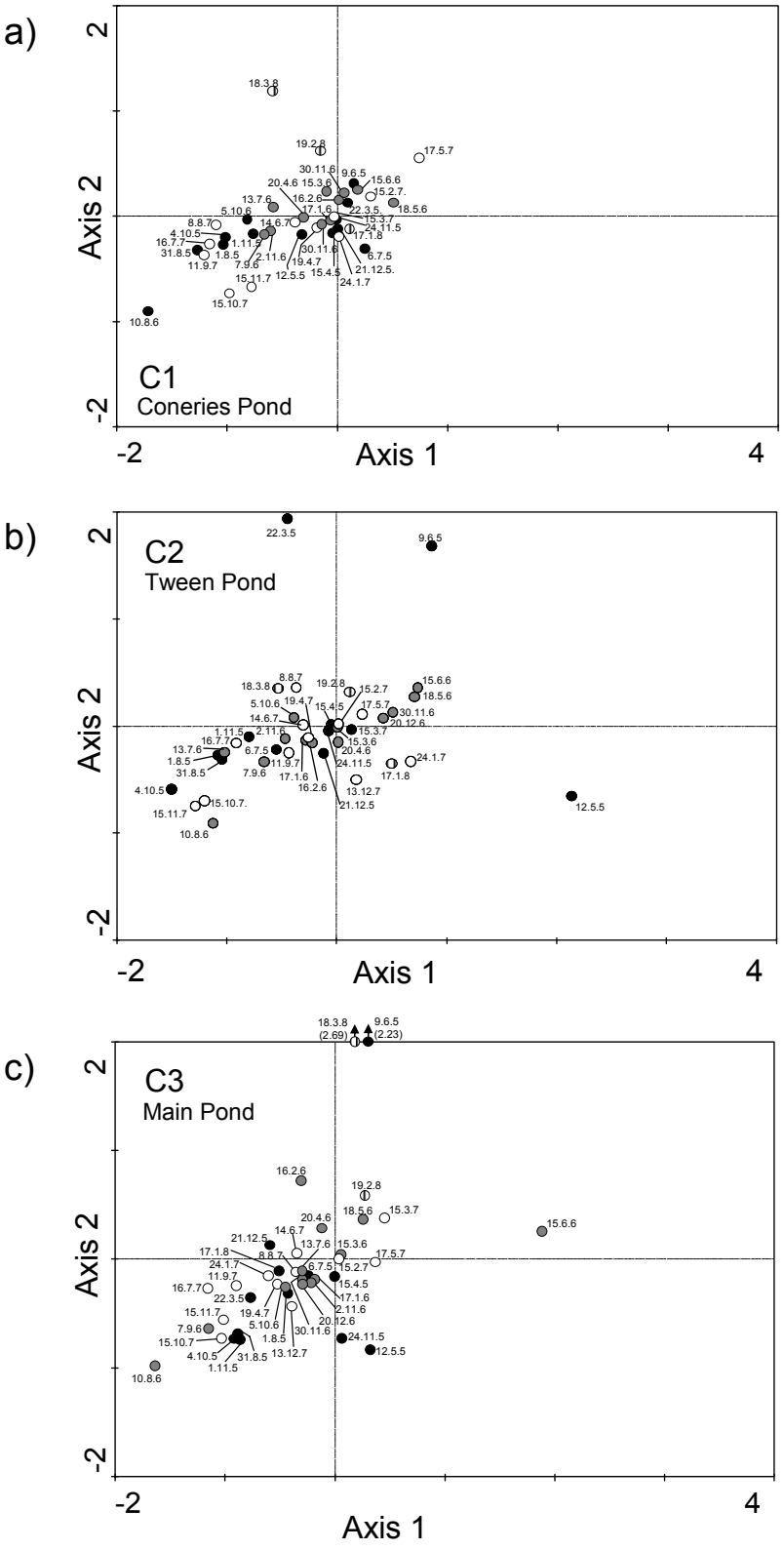

Figure 3.50 CA plots of phytoplankton samples from connected lakes, March 2005 to March 2008, a) Lake C1; b) Lake C2 and c) Lake C3; scaled by inter-sample distance using Hill's scaling. Samples are labelled by date (d.m.y) and colour-coded by year: black, 2005; grey, 2006; white, 2007; striped, 2008.

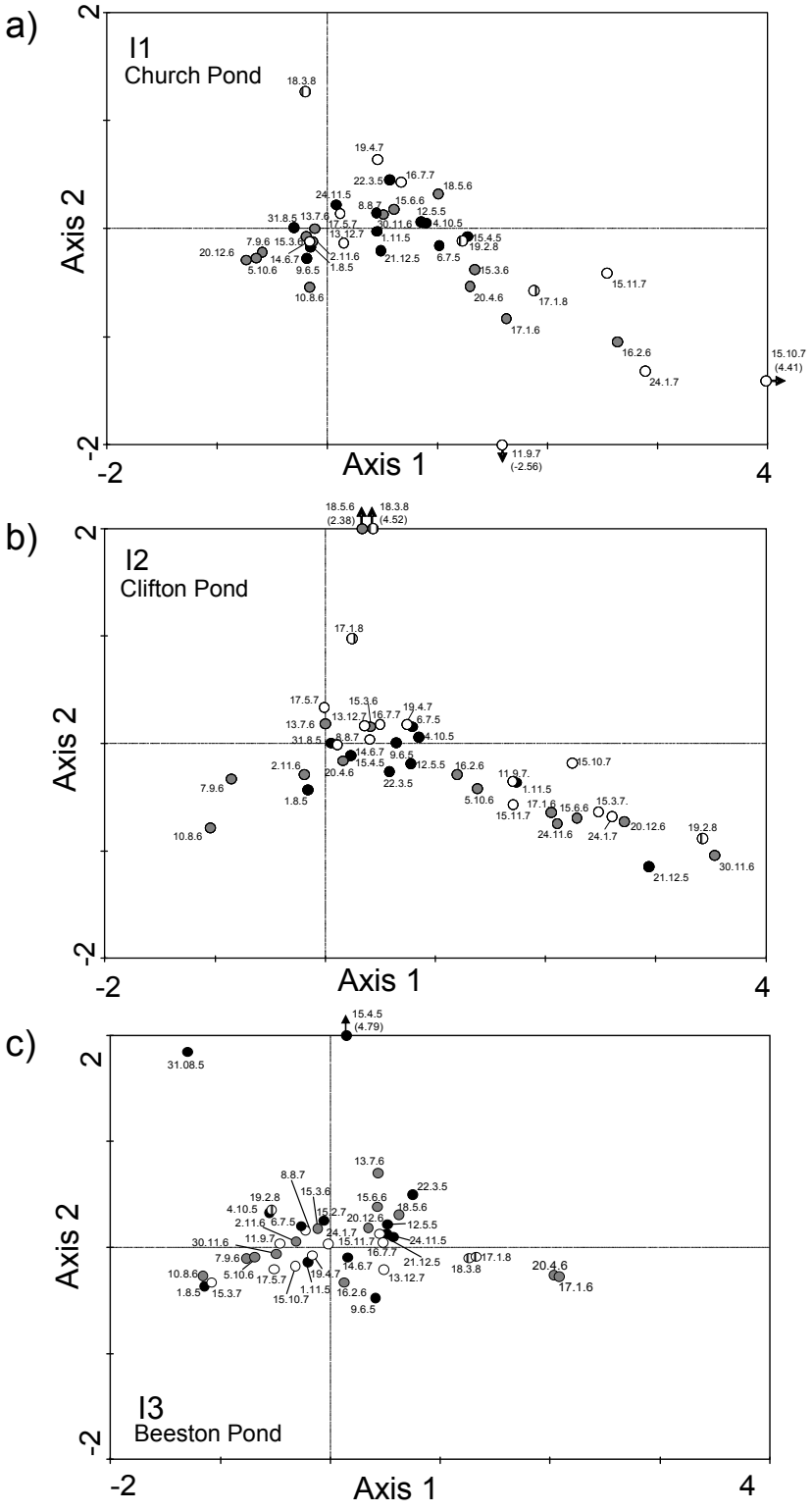

Figure 3.51 CA plots of phytoplankton samples from isolated lakes, March 2005 to March 2008, a) Lake I1; b) Lake I2 and c) Lake I3; scaled by inter-sample distance using Hill's scaling. Samples are labelled by date (d.m.y) and colour-coded by year: black, 2005; grey, 2006; white, 2007; striped, 2008.

the samples with the highest axis 1 scores were generally winter samples particularly during 2007. Lake I2 winter samples also tended to have high axis 1 scores. During winter, pennate diatoms were frequently observed in the phytoplankton of both lakes. In both lakes I1 and I2 cryptophytes were a significant component of phytoplankton community during winter, particularly during 2007. In Lake I1, the October 2007 sample was an outlier, due an isolated observation of *Dinobryon*. In Lake I2, samples from May 2006 and March 2008 were both outliers. During May 2006, small green flagellates were abundant, and in March 2008 *Gymnodinium* was abundant. Lake I3 sample scores (Figure 3.51c) were similar those of connected lakes. Samples in Lake I3 were not clearly separated by seasonality or differences between years. The majority of samples had low axis 1 scores, reflecting the dominance of diatoms and chlorophytes in the phytoplankton community. Late summer 2006 samples grouped towards the left of the CA plot due to the presence of *Microcystis* sp. in the phytoplankton. Samples from April 2005 and August 2005 were both outliers from the majority of samples because of the presence of abundant small green flagellates and *Gymnodinium* respectively.

Zooplankton community structure

Axes 1 and 2 identified by CA accounted for 14.7% and 14.1% of correlation between zooplankton sample and species scores, and had eigenvalues of 0.422 and 0.405 respectively. The first four axes cumulatively accounted for 54.4% of sample and species scores correlation, and total inertia was 2.875.

A CA of the zooplankton community (Figure 3.52) across all lakes revealed two clusters of taxa. Cyclopoid copepods, *Ceriodaphnia* sp., *Bosmina* spp. co-occurred with the rotifer *Asplancha* sp. and *D. longispina*, although *D. longispina* was rarely found in the zooplankton community. A second group of *D. pulex*, *Chydorus* sp. and calanoid copepods were associated with lower axis 1 and higher axis 2 scores. *D. hyalina* was found throughout all lakes and plots in between the two clusters. *D. cucullata* and the rotifer *Keratella* sp. were separate from both groups and were rarely found in any lake. Plant-associated zooplankton genera such as *Sida* sp. and *Alona* sp. formed a cluster towards the centre of the CA plot, although *Chydorus* sp. was not associated with other members of the Chydoridae.

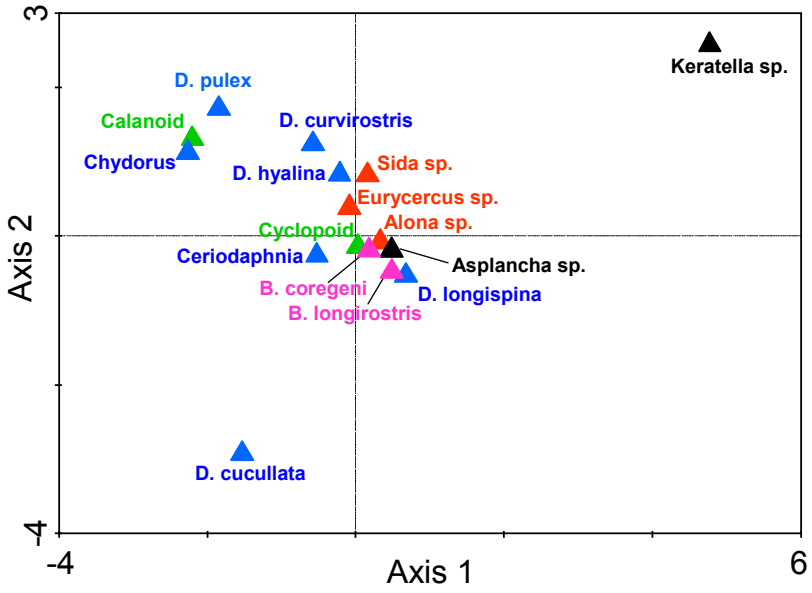

Figure 3.52 CA plot of zooplankton genera and species from all lakes, March 2005 to March 2008, scaled by inter-sample distance using Hill's scaling. Zooplankton genera and species are colour coded by group: blue, Daphnidae; green, copepoda; magenta, Bosminidae; rotifers, black; Chydoridae, red.

In connected lakes, particularly Lake C1 (Figure 3.53a), CA plots of sample scores show a split between early summer samples and other samples. In Lake C1, samples from May of all years had higher axis 1 and lower axis 2 scores. This is associated with the high abundances of *D. hyalina* observed during early summer. Samples from November and December 2006 plotted close to early summer samples, reflecting that *D. hyalina* was a dominant member of the wintertime zooplankton community during 2006. Unlike June 2005 and 2006 the June 2007 sample scored negatively on both axis 1 and axis 2 as a result of the increase in rotifers (particularly *Asplancha* sp.) observed during June 2007. During August 2008 rotifers remained relatively abundant which is reflected by the position of the sample towards *Keratella* sp. towards the right of Figure 3.52. The position of the majority of other samples indicate the dominance of cyclopoid copepods in the zooplankton community. A similar pattern was observed in Lakes C2 and C3. Samples from May and June 2005 and 2006, and November 2006 scored less on axis 1 and more highly on axis 2, also due to the relative dominance of *D. hyalina*. Samples from August 2008 were outliers in both lakes C2 and C3 as a consequence of the increase in rotifer abundance (mainly *Keratella* sp.). Samples from January 2007, outliers in both lakes C2 and C3 were almost entirely composed of *D. hyalina*.

Sample scores in lakes I1 and I2 (Figure 3.54a and b) were generally lower on axis 1 than in connected lakes, which can be explained due to higher number of calanoid copepods observed in lakes I1 and I2 compared to connected lakes. Seasonality in the abundance of zooplankton abundance was not clearly evident in lakes I1 and I2, which is reflected in the grouping of sample scores into one cluster. In Lake I1, samples from 2007 tended to cluster towards the right of the plot although interannual differences were not clear in Lake I2. The outlier from September 2006 is explained by the presence of *D. cucullata* in the zooplankton community.

Sample scores in Lake I3 were clustered towards the centre of the plot (Figure 3.54c) implying that the zooplankton community was most closely associated with cyclopoid copepods. As in isolated lakes I1 and I3, little seasonality was evident in zooplankton community structure, and interannual differences were not clearly visible. During 2007, rotifers were more abundant, evident in the positioning of summer 2007 samples close to *Asplancha* sp.

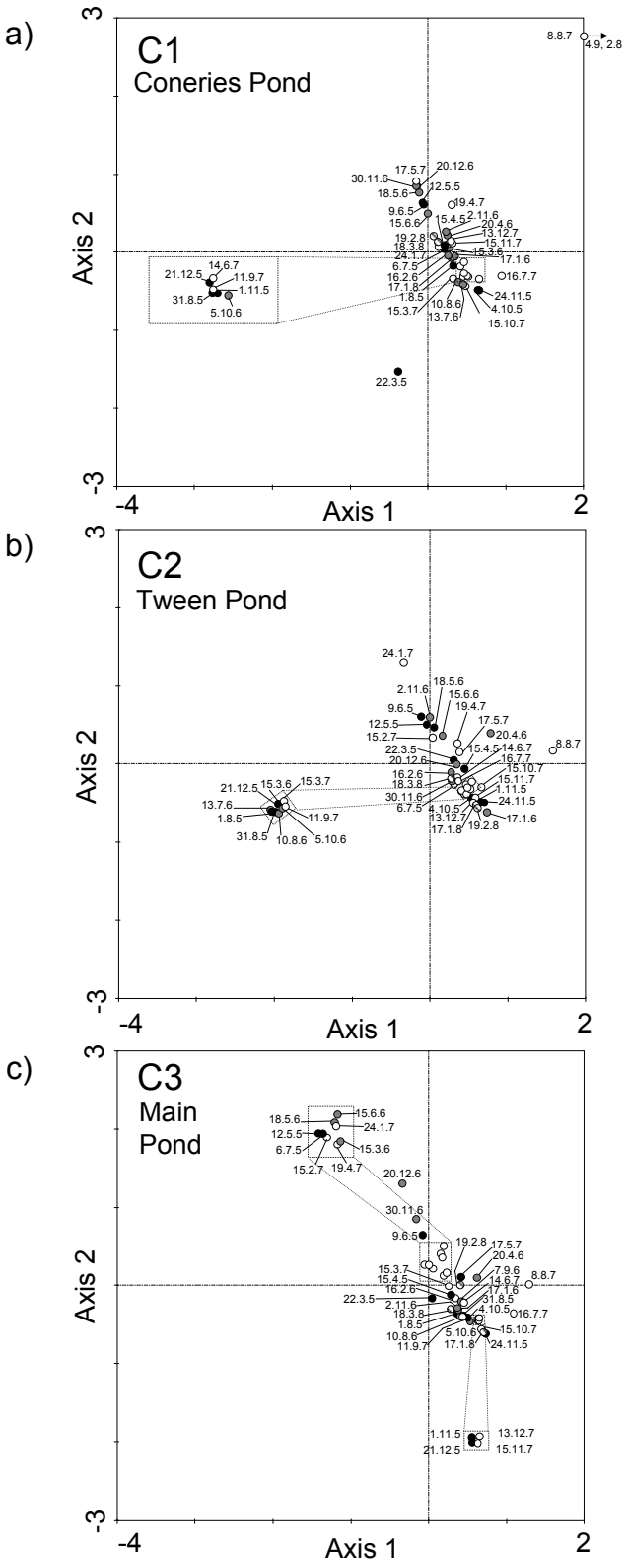

Figure 3.53 CA plots of zooplankton samples from connected lakes, March 2005 to March 2008, a) Lake C1; b) Lake C2 and c) Lake C3; scaled by inter-sample distance using Hill's scaling. Samples are labelled by date (d.m.y) and colour-coded by year: black, 2005; grey, 2006; white, 2007; striped, 2008.

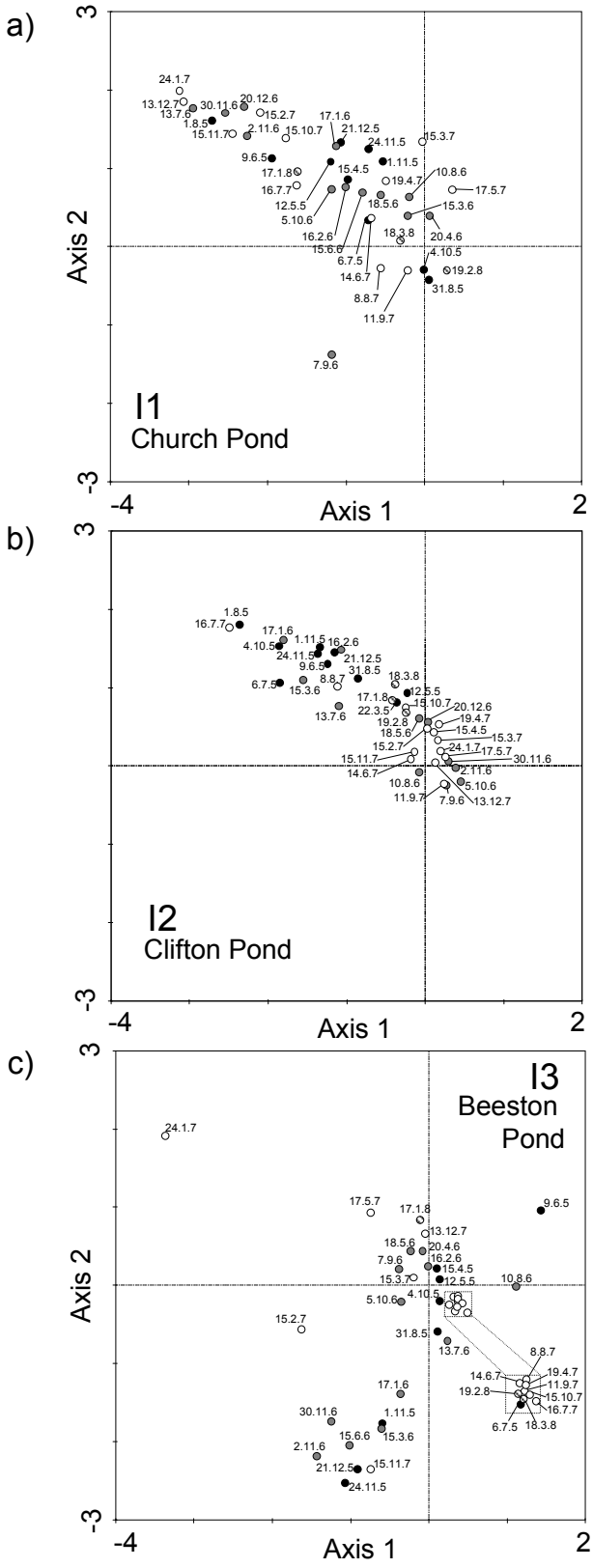

Figure 3.54 CA plots of zooplankton samples from isolated lakes, March 2005 to March 2008, a, Lake I1; b, Lake I2 and c, Lake I3; scaled by inter-sample distance using Hill's scaling. Samples are labelled by date (d.m.y) and colour-coded by year: black, 2005; grey, 2006; white, 2007; striped, 2008.

3.9 Discussion

Concentrations of TP and SRP in connected lakes at Attenborough Nature Reserve were substantially higher than in many other lowland shallow lakes in the U.K., Denmark and Germany (e.g. Schmitt and Nixdorf, 1999; Köhler and Hoeg, 2000; Lau and Lane, 2002; Jeppesen et al., 2003) and are well in excess of typical 'hypertrophic' P concentrations defined by the Organisation for Economic Cooperation and Development (OECD, 1982 in Dodds, 2002). In isolated lakes, particularly Lake I2, mean P concentrations were less than in connected lakes, although still falling within OECD classifications (see Table 1.1, page 36) of eutrophic (Lake I2) and hypertrophic (lakes I1 and I3). Mean concentrations of NO_3-N were also higher in connected lakes than isolated lakes. In connected lakes, mean annual NO_3-N concentrations were comparable with some of the highest concentrations found in south-east England (Bennion and Smith, 2000), and maximum winter concentrations were substantially higher than in shallow lakes throughout the U.K. and Poland (James et al., 2005). Mean NO_3-N concentrations in isolated lakes I1 and I2 fall below those observed by Bennion and Smith (2000) although are within the highest 50% of lakes with submerged vegetation presented by James et al. (2005). Mean NH_4-N concentrations in all isolated lakes were similar to those observed in Little Mere, U.K., after the diversion of sewage effluent (Beklioglu et al., 1999) and in Barton Broad, U.K. (Lau and Lane, 2002). In connected lakes values were over twice as high. Conductivity, major ions and Si concentrations were higher in connected lakes than isolated lakes. Conductivity and Si concentrations in connected lakes were higher than the majority surveyed by Bennion and Smith (2000). Higher Si concentrations were observed in Barton Broad by Lau and Lane (2002). Isolated lakes had higher conductivity and comparable Si concentrations to lakes in the Bennion and Smith (2000) survey. pH was similar or more alkaline compared to lakes in the south-east of England (Bennion and Smith, 2000) and Poland (Glińska-Lewczuk, 2009). High pH and conductivity are typical of lakes in calcareous catchments (e.g. Müller et al., 1998). In lowland areas, agricultural and urban land use is often higher, which can increase the release of nutrients into surface waters. High nutrient concentrations are not unexpected in the Attenborough Nature Reserve. Several STWs discharge into the River Erewash (see

section 2.1) which suggests that urbanisation may be particularly associated with the high nutrient concentrations.

The concentrations of major ions were generally higher in all connected lakes than isolated lakes, which suggests that the River Erewash was probably associated with this difference. Major ions are derived from weathering in catchments (Müller et al., 1998) and the River Erewash is likely to have been important for the transport of weathered material. As major ions are not mediated strongly by biological processes (Wetzel, 1983; Krivtsov and Sigee, 2005), biological differences between the connected and isolated lakes are unlikely to have been important in determining the concentrations of major ions, which supports the importance of the inputs from the River Erewash. Significant decreases in concentrations over time were observed in connected lakes but not in isolated lakes, which may reflect a transition towards more lotic conditions during 2007 because of dilution. Reductions in the concentrations of major ions over a gradient of increasing hydraulic connectivity have been observed by Glińska-Lewczuk (2009). Some decreases in major ion concentrations occurred during early 2008 in lakes I1 and I2, which may be associated with the increases in water depth and flooding that occurred during winter 2007-2008.

Strong seasonality was observed in the concentrations of nutrients in connected lakes. In both connected and isolated lakes, timeseries data showed that P concentrations increased during the late summer, which suggests that internal release from sediments is an important mechanism in determining late summer P concentration. For example, SRP concentrations increased during late summer 2005 and 2006 in Lake C3 (Figure 3.18, page 120) and Lake I2 (Figure 3.34a, page 147). Such patterns are typical in eutrophic lakes (e.g. Krivtsov et al., 2001; Reynolds and Davies, 2001; Tallberg and Koski-Vähälä, 2001; Spears et al., 2006). In highly productive lakes, oxygen consumption by bacterial decomposition of organic matter on the sediment surface causes anoxic conditions and allows P to be solubilised and released from the sediment (Moss et al., 1996a; Scheffer, 1998). In Danish lakes, Søndergaard et al. (2001) showed that in lakes with mean summer TP of >100 μg L^{-1}, late summer TP increased to a maximum of ~200–300% of winter TP concentrations. The highest increases in Attenborough Nature Reserve were comparable to this. For example, in

Lake C3 summer TP concentrations during 2006 were 265% of mean winter (January to March) TP concentrations.

NO_3-N and NH_4-N concentrations were higher in connected lakes than isolated lakes, reflecting the input of N from the River Erewash. NO_3-N and NH_4-N concentrations were also higher in Lake I3 than in the other isolated lakes, suggesting that the inflow to Lake I3 supplied N to the lake. The lack of N inputs into isolated lakes I1 and I2 and undetectable concentrations during the summer suggest the potential for N limitation of phytoplankton during the summer. A decreasing gradient in NO_3-N concentration from Lake C1 to lakes C2 and C3 was found. Denitrification (the conversion of NO_3-N to gaseous N_2, Wetzel, 1983) may contribute to this decline in water-column concentration. As denitrification depends on residence time (Scheffer, 1998), lakes further from the River Erewash inflow will have undergone additional denitrification compared to Lake C1 (see Figure 2.1, page 74) and therefore result in reduced water-column NO_3-N concentrations.

NO_3-N and NH_4-N concentrations generally decreased in all the lakes of Attenborough Nature Reserve during the summer. Nitrogen depletion during summer is a common feature of temperate lakes (e.g. Moss et al., 1994; Bennion and Smith, 2000; James et al., 2003). NH_4-N is the preferential form of N for uptake by macrophytes and phytoplankton (as NO_3-N and NO_2-N require reduction before uptake; Bronmark and Hannson, 1998). In connected lakes, NH_4-N would primarily be utilised by the phytoplankton community until reaching limiting levels, when NO_3-N may become a more important source of N. In the isolated lakes I1 and I2 where submerged macrophytes were abundant, N uptake by macrophytes probably contributed to the low summertime NO_3-N and NH_4-N concentrations.

Internal cycling of NH_4-N may have been important in determining water column concentrations, particularly in isolated lakes where there was no significant external input of NH_4-N. Zooplankton excretion and release from the sediments are likely to have been important processes. In all lakes, NH_4-N concentrations tended to increase after the summer chlorophyll-*a* maxima (e.g. Figure 3.10b, page 107), probably as a result of the decomposition of organic matter. The release of NH_4-N from sediments can contribute significantly to water column concentrations. For example, in Lake

Balaton, Hungary, Présing et al (2008) suggested that internal regeneration of NH_4-N was the principal source for phytoplankton. Beutel (2006) used experiments to demonstrate that NH_4-N release rates increased with eutrophication, which suggests that NH_4-N release may be comparatively more important in the connected lakes than the isolated lakes. Van Luijn (1999) found that release of NH_4-N release from sediment cores was much higher than NO_3-N and NO_2-N immediately after the spring phytoplankton bloom. In connected lakes, additional peaks of NH_4-N were observed in early summer. This is most likely to be a consequence of zooplankton excretion of NH_4-N (e.g. Wen and Peters, 1994; Attayde and Hansson, 1999; Vanni, 2002) during the spring clear-water phase. High abundances of *Daphnia* spp. observed during spring support the importance of zooplankton excretion.

Bergin and Hamilton (2007) suggest that NO_3-N loss by denitrification may be overestimated and that other processes can account for the fate of NO_3-N. For example, NO_3-N may also be reduced by dissimilatory nitrate reduction to ammonium (DNRA), potentially returning N in the form of NH_4-N, to be released to the water column (Bergin and Hamilton, 2007), which may be an important mechanism in reducing the degree of N limitation. In isolated lakes I1 and I2, comparatively low water-column concentrations on NH_4-N and NO_3-N can be explained by sedimentary conditions that were likely to be conducive for coupled nitrification and denitrification, for example by the release of oxygen around the roots of macrophytes during the summer months (Ottosen et al., 1999; Saunders and Kalff, 2001a) and lesser oxygen demand by mineralizing bacteria due to lower organic sedimentation (e.g. van Luijn et al., 1999). Therefore, both oxic and anoxic 'microenvironments' in isolated lakes are likely to exist, and consequently sedimentary conditions are ideal for coupled nitrification-dentrification (van Luijn et al., 1999; Beutel, 2006) which may therefore result in N limitation of phytoplankton growth (Seitzinger, 1988).

Strong seasonality was also observed in SiO_3 concentrations in connected lakes and Lake I3. Very rapid increases and decreases in SiO_3 concentration occurred in the connected lakes (e.g. Figure 3.2d, page 94). The seasonality of Si concentration in connected lakes corresponds to that observed in many other lakes and aquatic environments (e.g. Wang and Evans, 1969; Bailey-Watts, 1976; Gibson et al., 2000), driven by the uptake of Si by planktonic diatoms during the spring and late summer,

followed by the burial of diatom frustules in the sediments (e.g. Schelske and Stoermer, 1971; Schelske et al., 1983; Goto et al., 2007). Benthic diatoms may be more important than planktonic diatoms in determining Si concentrations in clear-water lakes such as lakes I1 and I2, due to improved benthic light climates caused by reduced pelagic production (e.g. Vadeboncoeur et al., 2001; Talling and Parker, 2002). Recycling of Si from the sediments is important in maintaining water column Si concentrations, particularly in isolated lakes where no significant external inputs exist. For example, Gibson et al. (2000) suggest that internal release contributed two-thirds of the Si required by diatoms for the spring bloom in Lough Neagh, and Parker et al. (1977) showed that less than 5% of silica required for annual diatom production was derived from catchment inputs in Lake Michigan. In the isolated lakes, internal release is likely to be important in maintaining water-column Si concentrations, although some authors have observed that groundwater inputs can be important sources of Si (e.g. Miretzky and Cirelli, 2004). Increases in Si concentrations were observed during autumn and winter 2006 in both lakes I1 and I2, following high phytoplankton productivity in the summer. The uptake of CO_2 by phytoplankton photosynthesis would cause an increase in pH (Brönmark and Hansson, 1998), which has been experimentally demonstrated to cause an increase in Si release (Lewin, 1961; Rippey, 1977, 1983). Although Si release and diatom dissolution may be driven by other processes, such as temperature (Rippey, 1977, 1983), salinity (Parker et al., 1977; Ryves et al., 2006) and physical processes (e.g. Flower, 1993), no discernible changes in these occurred at Attenborough Nature Reserve that would otherwise explain enhanced Si release.

In isolated lakes, mean chlorophyll-*a* concentrations were lower than in connected lakes. Under the OECD classification, lakes I1 and I2 fall into the 'eutrophic' category and I3 and connected lakes into the 'hypertrophic' category. The chlorophyll-*a* concentrations observed in connected lakes were similar to those of other shallow, lowland lakes with short retention times in western Europe, for example Lake Søbygård, Denmark, (Jeppesen et al., 1998), Barton Broad, U.K. (Lau and Lane, 2002) and lakes of the Scharmützelsee area of eastern Germany (Nixdorf and Deneke, 1997; Schmitt and Nixdorf, 1999). Maximum chlorophyll-*a* concentrations in connected lakes were comparable to some of the highest observed in south-east England by Bennion and Smith (2000) and were approximately three times higher

than those observed in Lake C2 during 1993 (Sayer and Roberts, 2001). Connected lakes were also highly turbid, shown by the low Secchi disk depths during summer (<0.4 m) and high total suspended solids concentrations. This suggests that phytoplankton probably contributed to turbidity during the summer and minerogenic turbidity was greater during the winter in connected lakes. The clearer water in isolated lakes I1 and I2 may in part be related to the absence of a source of minerogenic turbidity and implies that phytoplankton were the main source of turbidity in those lakes. Secchi depths similar to those of the connected lakes have been observed in other hypertrophic lakes, for example in Crest Lake, Louisiana (Burden et al., 1987) and in a dataset of 178 shallow lakes (Jensen et al., 1994), and shows no substantial change since Sayer and Roberts (2001) reporting of Secchi depths during 1993 in Lake C2. Mean concentrations of chlorophyll-*a* in lakes I1 and I2 are comparable to those of other macrophyte-dominated lakes, for example Little Mere, north-west U.K, after the diversion of sewage effluent (Moss et al., 1997), and Lake Krankesjön, Sweden (Blindow et al., 2002).

There were clear differences in phytoplankton community structure between isolated and connected lakes, but Lake I3 was more similar to the connected lakes than Lakes I1 and I2. The phytoplankton community composition of connected lakes during the monitoring period was dominated by diatoms, chlorophytes and cryptophytes. According to the classification of Reynolds et al. (2002) and Reynolds (2006), the phytoplankton community of connected lakes corresponds to groups J, X1 and D, considered typical of shallow and well-mixed water columns, and groups X1 and D are typical of turbid waters. All groups are considered to be sensitive to nutrient depletion. Genera indicative of these groups include *Scenedesmus*, *Coelastrum* and *Pediastrum* (J), *Chlorella*, *Ankyra*, and *Monoraphidium* (X1) and *Synedra*, *Nitzschia* and *Stephanodiscus* (D). In similar eutrophic shallow lakes, cyanobacteria often comprise a greater proportion of the phytoplankton community, such as in Lakes Melangsee (Schmitt and Nixdorf, 1999) and Müggelsee (Köhler and Hoeg, 2000), Germany. Jensen et al. (1994) suggested that in hypertrophic conditions, chlorophytes may dominate over cyanobacteria as their higher growth rates are able to exploit the abundance of nutrients more rapidly compared to cyanobacteria. Small rapidly reproducing species may also be favoured because of the high flushing rate in the connected lakes (Reynolds, 2006). In isolated lakes I1 and I2 cyanobacteria formed a

higher percentage of the phytoplankton community than in connected lakes and cryptophytes were more abundant than diatoms. The phytoplankton community of Lake I1 included genera representative of groups X1 and J, H1 (*Anabaena* and *Aphanizomenon*), and G (*Eudorina* types, particularly during 2006). In Lake I2, groups X1, H1, Y (*Cryptomonas*, *Peridinium*) and Lm (including *Ceratium*, during 2006 particularly) were represented during the monitoring period. The phytoplankton community of both lakes appears to be related to moderately eutrophic conditions with evidence of N limitation and a sensitivity to low light. Lake I3 was dominated by cryptophytes and diatoms during the monitoring period. In addition to genera representative of the groups X1 and Y and J, groups B and C (principally *Aulacoseira* and including *Asterionella*) and W1 and W2 (*Euglena* and *Trachelomonas* type) indicate a community typical of small, mixed and eutrophic ponds (Reynolds et al., 2002; Reynolds, 2006). The general absence of cyanobacteria and abundance of diatoms suggests the phytoplankton community of Lake I3 was more similar to that of the connected lakes than the isolated lakes.

Abundant submerged vegetation and clear water was found in lakes I1 and I2, although in Lake I3 and connected lakes submerged macrophytes were rare and the water turbid. The vegetation communities of lakes I1 and I2 are typical of eutrophic lowland standing water. Lake I1 largely corresponds to the A12 *Potamogenton pectinatus* community (Rodwell et al., 1995). The community is known to flourish in naturally or artificially enriched water and is tolerant of turbid water. In Lake I2, the submerged vegetation community was more diverse, and similar to the A11 *P. pectinatus* - *Myriophyllum spicatum* community (Rodwell et al., 1995), which is considered typical of lowland, mesotrophic to eutrophic water.

Zooplankton abundance differed among connected and isolated lakes. In connected lakes and Lake I3, zooplankton abundance frequently reached approximately 40 ind. L^{-1}, although in lakes I1 and I2 zooplankton abundance rarely exceeded 20 ind. L^{-1}. One potential explanation for this is that the phytoplankton community of connected lakes was dominated by species susceptible to grazing (Reynolds, 2006), therefore providing an abundant food source for the grazing community (e.g. Elser and Goldman, 1991) in connected lakes. In connected lakes, *D. hyalina* were more abundant during early summer clear-water periods. Similar abundances of *Daphnia*

have been measured in other eutrophic shallow lakes (e.g. Timms and Moss, 1984; Jacobsen and Simonsen, 1993; Deneke and Nixdorf, 1999). Cyclopoid copepods and *Bosmina* sp. were abundant in connected lakes during summer, a pattern which has also been observed in Hoveton Great Broad (Timms and Moss, 1984). Zooplankton in isolated lakes I1 and I2 was more closely associated with calanoid copepods, and *Daphnia* spp. were not just restricted to the spring clear-water phase. The occurrence of larger *Daphnia*, particularly in Lake I2, suggests that the abundance of submerged macrophytes may be important in providing a refuge for zooplankton against zooplanktivorous fish (e.g. Timms and Moss, 1984; Scheffer, 1998). Large zooplanktivorous fish have been observed in both isolated lakes I1 and I2 and in the connected lakes, which suggests that predation on zooplankton may be important in determining their abundance. The strength of zooplankton grazing as a limiting factor for phytoplankton growth is explored further in Chapter 4.

Both clear and turbid states (Moss et al., 1996a; Scheffer and Jeppesen, 1998; Scheffer, 2001) are represented in the Attenborough Nature Reserve. The low total suspended solids concentrations, low nutrient concentrations and deep Secchi disk depths observed in lakes I1 and I2 are consistent with this theory. In connected lakes and Lake I3, the water was turbid, higher nutrient concentrations existed, there was a lack of submerged vegetation and high chlorophyll-*a* concentrations which represents typical conditions of the turbid state. Sayer and Roberts (2001) suggested that the diversion of the River Erewash into lakes C1, C2 and C3 caused a switch from clear water to turbid conditions. The data presented in this chapter supports the suggestion that the connected lakes are in a turbid state.

During 2007 distinct differences in the chemistry and biology were observed compared to the two previous years, particularly in the connected lakes. Maximum P concentrations were less than 2005 and 2006, and P concentrations declined more slowly during the growth season in both connected and isolated lakes. NH_4-N concentrations were below detectable limits for longer during the summer of 2007 than previous years in the connected lakes. Maximum chlorophyll-*a* concentrations were less than half of that observed during 2005 and 2006 in connected and isolated lakes. Late summer *Daphnia* populations were also lower during 2007 in the connected lakes. The reasons for the differences appear to be related to high rainfall and

consequent changes in the hydrology of connected lakes. However, the effects of extreme short-term hydrological events on shallow lake functioning have not been widely investigated. A reduction in the importance of internal processes, such as sedimentary nutrient release and grazing, and a switch towards external processes dominating in-lake nutrient concentrations is a possible explanation (e.g. Angeler et al., 2000). A reduction of chlorophyll-*a* concentrations with flushing is supported by evidence from theoretical (Reynolds, 2006) and modelling perspectives (e.g. Jones and Elliot, 2007; Elliott et al., 2009). Empirical evidence of this in shallow lakes is lacking, although some authors have found that high-flushing conditions reduce chlorophyll-*a* concentrations (Paidere et al., 2007; Haldna et al., 2008) and favours the growth of small, rapidly growing groups, particularly cryptophytes and *r*-strategist diatoms (Huszar and Reynolds, 1997; Schmitt and Nixdorf, 1999; Paidere et al., 2007). The importance of these processes and observations is analysed in more depth in Chapter 5.

3.10 Conclusion

Evidence presented in this chapter indicates that connectivity to the nutrient-rich River Erewash is associated with substantial differences in the lakes of Attenborough Nature Reserve. Contrasts were observed in the availability of nutrients, community structure and composition of phytoplankton and zooplankton. Lake I3, although isolated from the River Erewash had substantially higher nutrient concentrations and different phytoplankton, zooplankton and submerged vegetation communities to isolated lakes I1 and I2, probably because of a nutrient-rich inflowing stream.

Connectivity to the River Erewash resulted in substantially greater concentrations of major nutrients in connected lakes than in isolated lakes. All lakes of Attenborough Nature Reserve can be categorised as eutrophic, but connected lakes and Lake I3 are representative of hypertrophic conditions. Evidence of internal processes, including the recycling of N, P and Si from the sediments, existed in all lakes and are probably important in determining in-lake nutrient concentrations. Internal mechanisms are more important in the isolated lakes than connected lakes as they are not subjected to high external loading of nutrients.

Phytoplankton communities differed between connected and isolated lakes. Connected lake phytoplankton communities comprised smaller and more rapidly reproducing species, probably in order to exploit the abundance of nutrients and low WRT. Diatoms in particular were substantially more abundant in connected lakes and were associated with rapid changes in Si concentrations. The prolonged periods of low N concentrations in isolated lakes I1 and I2, in addition to the lack of flushing, were associated with cyanobacteria being relatively more abundant. The potential effects of the interactions between zooplankton grazing and phytoplankton communities are addressed in Chapter 4.

Preliminary assessments of the timescales of nutrient, phytoplankton and zooplankton datasets in the lakes of Attenborough Nature Reserve in this chapter suggest that reductions in P concentrations, chlorophyll-*a* concentrations and zooplankton occurred during 2007. Shifts in phytoplankton community structure were also observed, with shifts away from cyanobacteria and towards diatoms and cryptophytes. The extent to which these changes may be associated with hydrological changes are explored in Chapter 5.

Chapter 4 Relationships between resources, ecosystem structure and function in Attenborough Nature Reserve

4.1 Introduction

This chapter attempts to determine how phytoplankton communities may be affected by resource availability, and how structure and functioning may vary among the lakes of Attenborough Nature Reserve. The data used in this chapter were discussed in detail in Chapter 3 and comprises data from all lakes of Attenborough Nature Reserve. The data presented in Chapter 3 suggested that nutrient concentrations in connected lakes were strongly affected by the River Erewash, and that zooplankton are abundant and could potentially affect total phytoplankton biomass. Evidence was presented to show that connectivity to the River Erewash was associated with a phytoplankton community dominated by small species that may be more susceptible to grazing, whereas isolated lakes I1 and I2 tended to be dominated by larger and possibly grazing-resistant species.

Controls of phytoplankton abundance and composition can vary between lakes with submerged vegetation and those without. In lakes with abundant submerged macrophytes, zooplankton may exert a strong control on phytoplankton communities because of the presence of refugia for zooplankton, and maintain top-down control even when the abundance of zooplanktivorous fish is high (e.g. Timms and Moss, 1984). Jeppesen et al. (1999; 2003) have suggested that the effects of zooplankton on phytoplankton probably increase with trophic state in lakes with macrophytes because of reduced bottom-up control. However, Blindow et al. (2000) found that when submerged vegetation is highly abundant, nutrient limitation of phytoplankton may have been more important than zooplankton grazing.

Many studies suggest that the effects of grazing and nutrient limitation on lake phytoplankton may vary temporally as well as between lakes. For example,

zooplankton grazing can be particularly intense in many eutrophic lakes during the spring, and as a result the biomass of small and easily-grazed phytoplankton species is substantially reduced (e.g. Vanni and Temte, 1990; Talling, 2003). Later in the growth season, the strength of top-down control is often reduced. Phytoplankton that are resistant to grazing reduce the strength of top-down control. Grazing resistance primarily arises because of the size and toxicity of cyanobacteria (Agrawal, 1998; Lampert and Sommer, 2007). Cyanobacteria can become more dominant after clear-water phases in eutrophic lakes, and may be dominant in hypertrophic lakes, resulting in a reduction of top-down phytoplankton control. Bottom-up control of phytoplankton may also become stronger as dissolved nutrients are assimilated into the phytoplankton during the summer in eutrophic lakes (Lampert and Sommer, 2007).

This chapter will use a correlative approach to determine if any correlations exist between nutrient and zooplankton variables among the lakes of Attenborough Nature Reserve from which casual mechanisms may be inferred. The following questions are addressed:

- Which biological and physico-chemical variables are correlated with chlorophyll-*a* concentrations?
- Which biological and physico-chemical variables are associated with changes in phytoplankton community structure?
- How do these relationships vary among connected and isolated lakes?

4.2 Methods

The relationship between chlorophyll-*a* concentrations and zooplankton, chemical and physico-chemical variables was assessed using Spearman's rank correlation (r_s). Spearman's rank correlation is a non-parametric form of correlation. It was chosen over parametric correlation approaches (such as the Pearson correlation coefficient) for this analysis as it avoided the need for transformations to be applied to environmental, physico-chemical and zooplankton datasets in order to meet the normal distribution assumptions of parametric correlation (Field, 2000). Using Spearman's rank correlation is also a more flexible analysis of the association between two variables, as it measures the consistency of a relationship rather than the

strength of a linear relationship as with the Pearson correlation coefficient (Gravetter and Wallnau, 2000). Spearman's rank correlations were computed using SPSS 15.

Canonical correspondence analysis (CCA) was then used in order to relate the phytoplankton community composition to biotic and abiotic variables which were transformed where required to conform to the assumption of normality (Table 4.1). Prior to performing CCA, DCA was used to assess the length of the first axis of the phytoplankton dataset, and suggested that a unimodal approach would be appropriate (Table 4.2). CCAs were performed using CANOCO 4.5 on phytoplankton species that were summed to genus level and $\log_{(x+1)}$ transformed. Genera that contributed less than 5% of the total biovolume of a sample were excluded from the analysis to reduce noise in the dataset. Scaling of ordination scores was focused on interspecies correlations in order that CCA plots would show the differences between the occurrence of different phytoplankton species and correlation amongst environmental variables. Since analysis was based on unimodal species response curves, Hill's scaling was used. In order to determine which environmental variables independently explained a significant proportion of species variability, forward selection with 1000 Monte Carlo permutations was used. Variables were considered significant if $p \leq 0.05$. CCA was then performed using the significant explanatory environmental variables alone, and both the significance of the first ordination axis and all canonical axes was assessed using a Monte Carlo test with 1000 permutations under a reduced model (Lepš and Šmilauer, 2003). Reduced models are recommended for small datasets as it reduces the likelihood of Type I errors (ter Braak and Šmilauer, 2002).

Table 4.1 Transformations applied to chemical, physico-chemical and zooplankton variables entered into CCAs by lake. 'sqrt', square root. Abbreviations used in CCA plots are given in parentheses.

Variable	Transformation applied by lake					
	C1	C2	C3	I1	I2	I3
Chemical variables						
TP	-	$\log_{(x+1)}$	$\log_{(x+1)}$	$\log_{(x+1)}$	-	$\log_{(x+1)}$
SRP	-	-		$\log_{(x+1)}$	$\log_{(x+1)}$	sqrt
NO_3-N	$\log_{(x+1)}$	$\log_{(x+1)}$	$\log_{(x+1)}$	$\sqrt[3]{}$	$\log_{(x+1)}$	sqrt
NH_4-N	sqrt	sqrt	sqrt	$\sqrt[3]{}$	-	$\sqrt[3]{}$
SiO_3	-	-	-	sqrt	$\sqrt[3]{}$	-
Total Alkalinity (Alk)	-	-	-	-	-	-
Cl^-	-	-	-	-	-	-
Na^+	-	-	-	-	-	-
K^+	-	-	-	-	-	-
Mg^{2+}	-	-	-	-	-	-
Ca^{2+}	-	-	-	-	-	-
Physico-chemical variables						
Temp (T)	sqrt	sqrt	sqrt	sqrt	sqrt	sqrt
pH	-	-	-	-	-	-
Cond	-	-	-	sqrt	-	-
Zooplankton groups						
Daphnia spp. (Dap)	$\log_{(x+1)}$	$\log_{(x+1)}$	$\log_{(x+1)}$	$\log_{(x+1)}$	$\log_{(x+1)}$	$\log_{(x+1)}$
Ceriodaphnia spp. (Cer)	$\log_{(x+1)}$	$\log_{(x+1)}$	$\log_{(x+1)}$	$\log_{(x+1)}$	$\log_{(x+1)}$	$\log_{(x+1)}$
Calanoid copepods (Cal)	$\log_{(x+1)}$	$\log_{(x+1)}$	$\log_{(x+1)}$	$\log_{(x+1)}$	$\log_{(x+1)}$	$\log_{(x+1)}$
Cyclopoid copepods (Cyc)	$\log_{(x+1)}$	$\log_{(x+1)}$	$\log_{(x+1)}$	$\log_{(x+1)}$	$\log_{(x+1)}$	$\log_{(x+1)}$
Bosmina spp. (Bos)	$\log_{(x+1)}$	$\log_{(x+1)}$	$\log_{(x+1)}$	$\log_{(x+1)}$	$\log_{(x+1)}$	$\log_{(x+1)}$

Table 4.2 Axis 1 lengths determined by DCA of phytoplankton data using detrending by segments.

Lake	Axis 1 length
C1	2.885
C2	4.244
C3	3.615
I1	3.907
I2	4.638
I3	3.931

4.3 Results

4.3.1 Lake C1 (Coneries Pond)

In Lake C1, positive correlations were found between chlorophyll-*a* concentrations and the zooplankton taxa cyclopoid copepods and *Bosmina* spp. (Table 4.3). Weaker

correlations existed between chlorophyll-*a* concentrations and temperature, pH and total alkalinity. All of the negative associations with chlorophyll-*a* concentration were nutrients. NH_4-N of NO_3-N were more strongly negatively correlated with chlorophyll-*a* concentrations than SiO_3 concentrations.

Table 4.3 Spearman's rank correlation coefficients between chlorophyll-*a* concentration and zooplankton and environmental variables in Lake C1 (Coneries Pond), March 2005-March 2008.

Variable	Spearman's rank correlation	
Cyclopoid	0.564	
Bosmina spp.	0.516	
Temperature	0.505	
pH	0.391	
Total alkalinity	0.377	
Mg^{2+}	0.331	
NH_4-N		-0.781
SiO_3		-0.530
NO_3-N		-0.326

CCA analysis revealed seven environmental variables which were independently and significantly (p <0.05, Monte Carlo test) correlated with variance in the phytoplankton dataset using forward selection. Both abiotic variables (NH_4-N, NO_3-N, Na^+,Mg^{2+}, and temperature) and biotic variables (*Daphnia* spp.) were significant explanatory variables. Axis 1 explained 9.2% of the variance in the phytoplankton community (Table 4.4) and was strongly correlated with NH_4-N and *Daphnia* spp. (r = 0.567 and 0.531 respectively). Axis 2, which explained an additional 6.4% of phytoplankton community variance, was strongly correlated with NO_3-N concentration (r = -0.806). Temperature was positively correlated with axis 2 (r = 0.450). Axes 3 and 4 were most strongly correlated with temperature and SiO_3 (r = -0.518 and -0.361 respectively) and together explained a further 10.1% of phytoplankton community variance.

Table 4.4 Summary of the results of CCA analysis between phytoplankton species and biotic and abiotic variables in Lake C1 (Coneries Pond), March 2005-March 2008. Significance values are derived from a 1000 permutation Monte Carlo test.

	Axis 1	Axis 2	Axis 3	Axis 4
Eigenvalue	0.315	0.217	0.200	0.145
Species - environment correlation	0.871	0.862	0.839	0.756
Cumulative percentage variance				
of species data	9.2	15.6	21.5	25.7
of species - environment relation	28.3	49.9	65.8	78.9
Sum of all eigenvalues	3.412			
Sum of all canonical eigenvalues	1.112			
Variance explained by CCA (%)	32.59			
	F	*p*		
Significance of first canonical axis	3.050	0.001		
Significance of all canonical axes	2.073	0.001		

The CCA plots (Figure 4.1a and b) show winter samples being more closely associated with high concentrations of NO_3-N, NH_4-N and SiO_3 reflecting the higher concentrations of these nutrients observed during the winter. Early summer samples were associated with *Daphnia* spp. reflecting frequent presence of *Daphnia* spp. in the zooplankton community during May and June. Samples from the summer of all years tended to occur on the left hand side of the plots, indicating their association with increased temperature, and higher Mg^{2+} concentrations, observed particularly during 2006 and 2007. The distribution of species in the CCA plot (Figure 4.1) shows species typically observed during the summer months plotting along the temperature gradient (such as the cyanobacteria *Microcystis* and *Oscillatoria*). Several genera plotted near the centre of the CCA plot, including *Cryptomonas* spp. and centric and pennate diatom species, which suggests their consistent presence throughout the monitoring period. *Ankyra* spp. was associated with *Daphnia* spp. and higher NH_4-N and SiO_3 concentrations which corresponds to the higher abundance of *Ankyra* spp. observed during early summer. Diatom taxa, other than centric species, were negatively associated with SiO_3 suggesting uptake by diatoms strongly influences seasonal variability in SiO_3 concentrations.

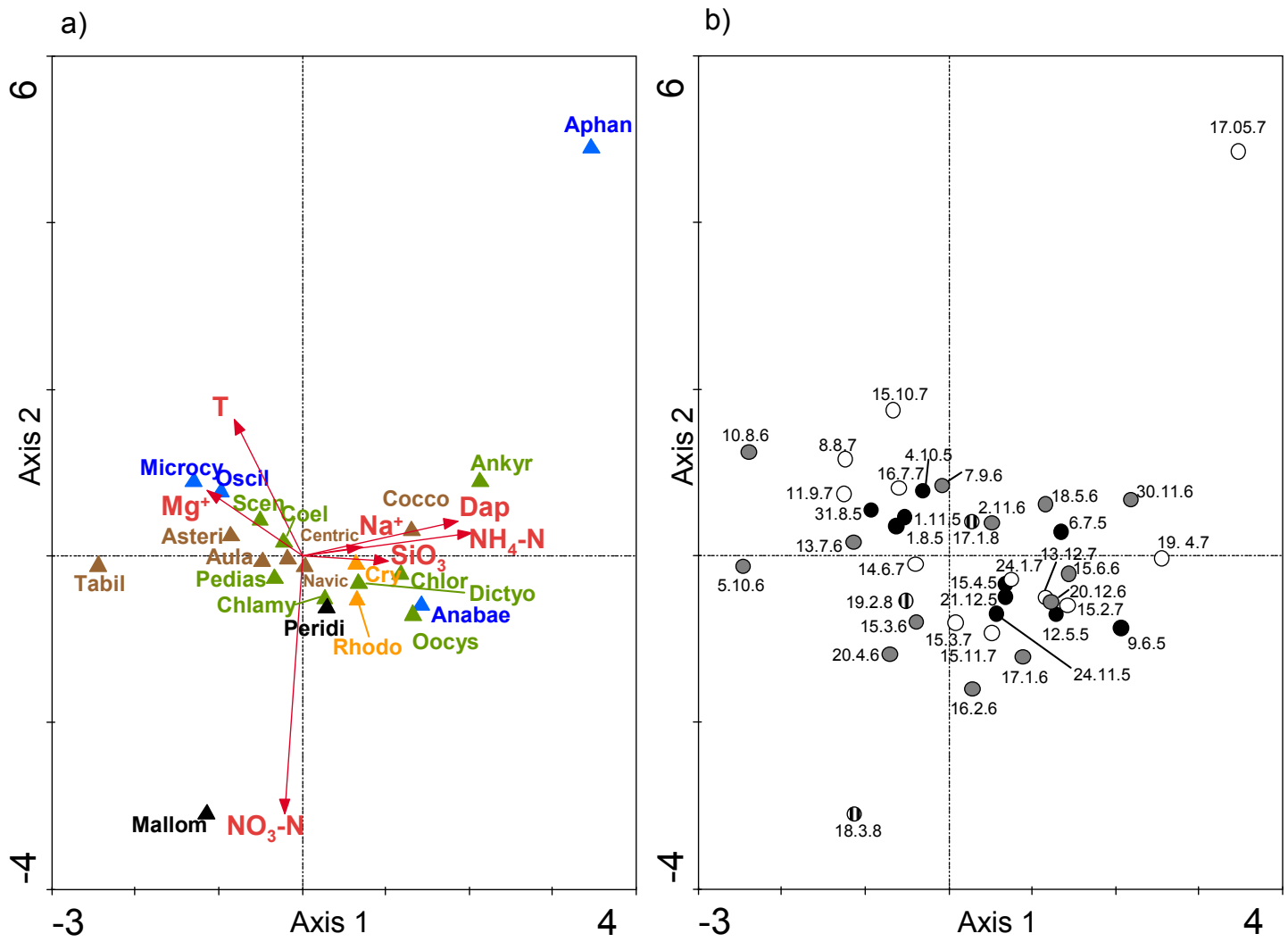

Figure 4.1 CCA plots of Lake C1 (Coneries Pond) constrained by the environmental variables. Only the first and second axes are shown. a) Phytoplankton species colour coded by phytoplankton group: brown, Bacillariophyceae; green, chlorophytes; orange, cryptophytes; blue, cyanobacteria; black, others (including chrysophytes and dinophytes). Phytoplankton abbreviations are given in Table 3.1, page 178 and environmental variable abbreviations in Table 4.1, page 200. b) samples, labelled by date (d.m.y) and colour-coded by year: black, 2005; grey, 2006; white, 2007; striped, 2008.

4.3.2 Lake C2 (Tween Pond)

Several positive correlations were found between chlorophyll-*a* concentration and environmental and zooplankton variables (Table 4.5). The strongest correlations with chlorophyll-*a* concentrations were found with pH and zooplankton groups. Chemical variables were comparatively more weakly positively associated with chlorophyll-*a* concentrations (r_s ≤0.585). Three chemical variables (NH_4-N, NO_3-N and SiO_3 concentrations) were strongly negatively associated with chlorophyll-*a* concentrations (Table 4.5).

Table 4.5 Spearman's rank correlation coefficients between chlorophyll-*a* concentration and zooplankton and environmental variables in Lake C2 (Tween Pond) March 2005-March 2008.

Variable	Spearman's rank correlation
pH	0.697
Cyclopoid	0.695
Rotifers	0.621
Total alkalinity	0.585
Bosmina spp.	0.523
Temperature	0.497
TP	0.480
Mg^{2+}	0.462
K^+	0.414
Na^+	0.337
NH_4-N	-0.792
SiO_3	-0.548
NO_3-N	-0.528

CCA analysis showed that seven variables were independently and significantly correlated with the phytoplankton dataset after forward selection. The first axis identified explained 10.2% of phytoplankton community variance (Table 4.6), and was most strongly correlated with *Bosmina* spp. (*r* = -0.605) and *Daphnia* spp. (*r* = 0.564). Axis 2, which explained a further 6.4% of phytoplankton community variance, was strongly positively correlated with calanoid copepods (*r* = 0.748). Axes 3 and 4 (explaining an additional 10.5% of phytoplankton community composition together) were most strongly correlated with *Ceriodaphnia* spp. (axis 3) and Ca^{2+} concentrations (axis 4). CCA showed positive correlations amongst zooplankton groups. *Bosmina* sp. was correlated with *Ceriodaphnia* sp. (*r* = 0.476), and *Daphnia* spp. with calanoid

copepods (*r* = 0.364). *Bosmina* sp. were also positively correlated with TP concentration.

Table 4.6 Summary of the results of CCA analysis between phytoplankton species and environmental variables and zooplankton abundance in Lake C2 (Tween Pond), March 2005 to March 2008. Significance values are derived from a 1000 permutation Monte Carlo test.

	Axis 1	Axis 2	Axis 3	Axis 4
Eigenvalue	0.408	0.256	0.240	0.177
Species - environment correlation	0.906	0.855	0.816	0.795
Cumulative percentage variance				
of species data	10.2	16.6	22.6	27.1
of species - environment relation	31.8	51.7	70.4	84.2
Sum of all eigenvalues	3.993			
Sum of all canonical eigenvalues	1.284			
Variance explained by CCA (%)	32.16			
	F	*p*		
Significance of first canonical axis	3.414	0.057		
Significance of all canonical axes	2.369	0.001		

CCA plots show summer phytoplankton genera and samples plotting towards the left-hand side of the plots (Figure 4.2a and b), including *Microcystis*, *Oscillatoria*, *Pediastrum* and *Scenedesmus*. Summertime samples tended to be associated with high abundance of *Bosmina* sp. and *Ceriodaphnia* spp. Spring and early summer samples tended to plot towards the right-hand side of Figure 4.2b, with increases in the biomass of *Ankyra* spp. associated with increased abundance of *Daphnia* sp. Other small chlorophytes including *Chlorella* spp. and *Chlamydomonas* spp. were associated with spring and early summer samples. Wintertime samples plotted towards the centre of Figure 4.2b suggesting an association with low zooplankton biomass and TP concentrations. Diatoms (except *Cocconeis* sp.) and cryptophytes plotted in the centre of the CCA plot, reflecting their presence in the phytoplankton community throughout the monitoring period.

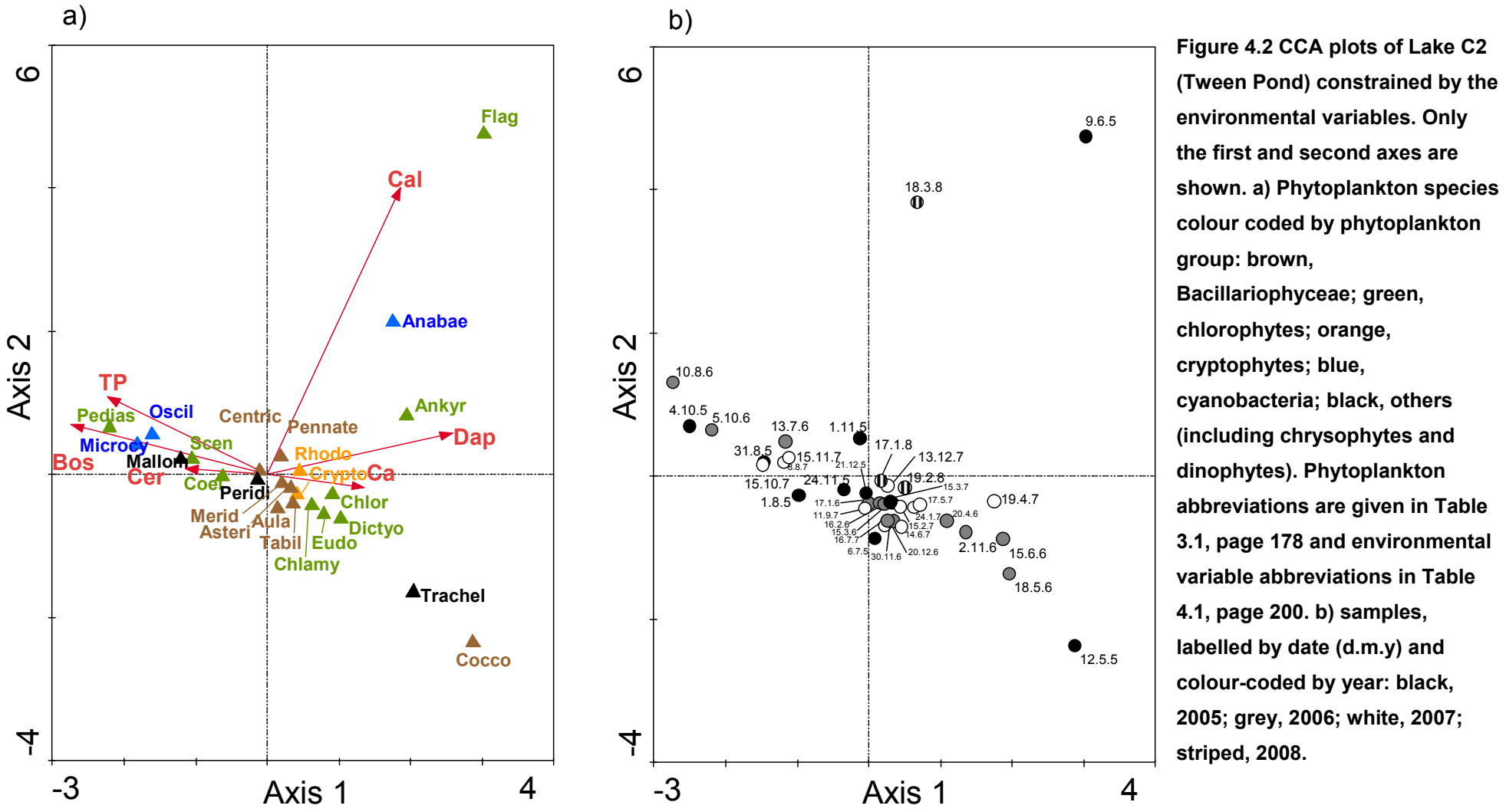

Figure 4.2 CCA plots of Lake C2 (Tween Pond) constrained by the environmental variables. Only the first and second axes are shown. a) Phytoplankton species colour coded by phytoplankton group: brown, Bacillariophyceae; green, chlorophytes; orange, cryptophytes; blue, cyanobacteria; black, others (including chrysophytes and dinophytes). Phytoplankton abbreviations are given in Table 3.1, page 178 and environmental variable abbreviations in Table 4.1, page 200. b) samples, labelled by date (d.m.y) and colour-coded by year: black, 2005; grey, 2006; white, 2007; striped, 2008.

4.3.3 Lake C3 (Main Pond)

Nine positive correlations between chemical, physico-chemical and zooplankton variables were found (Table 4.7). The strongest positive correlations existed between the abundance of cyclopoid copepods, pH and the abundance of *Bosmina* spp. Weaker positive correlations were found between major ions (Mg^{2+} and K^{+}), TP, water temperature and specific conductivity. The three negative correlates of chlorophyll-*a* concentration were with NH_4-N, NO_3-N and SiO_3 concentrations (Table 4.7).

Table 4.7 Spearman's rank correlation coefficients between chlorophyll-*a* concentration and zooplankton and environmental variables in Lake C3 (Main Pond) March 2005-March 2008.

Variable	Spearman's rank correlation
Cyclopoid	0.738
pH	0.649
Bosmina spp.	0.557
Temperature	0.538
Mg^{2+}	0.489
K^{+}	0.437
TP	0.376
Total alkalinity	0.368
Spec. cond.	0.334
NH_4-N	-0.706
NO_3-N	-0.613
SiO_3	-0.572

In Lake C3, CCA suggested that three of the environmental variables independently and significantly were correlated with variance in the phytoplankton dataset after forward selection. 8.2% of variance in the phytoplankton community was explained by axis 1 (Table 4.8), which was positively correlated with *Daphnia* spp (r = 0.63) and less strongly positively correlated with temperature (r = 0.26). *Bosmina* spp. was weakly negatively correlated with axis 1 (r = -0.15). Axis 2 explained an additional 5.5% of phytoplankton community variation and was positively correlated with all three environmental variables, most strongly *Bosmina* spp. and temperature (r = 0.76 and 0.43 respectively). Axis 3 explained 4.3% of the phytoplankton community variance and was strongly correlated with temperature (r = 0.62).

Table 4.8 Summary of the results of CCA analysis between phytoplankton species and environmental variables and zooplankton abundance in Lake C3 (Main Pond), March 2005 to March 2008. Significance values are derived from a 1000 permutation Monte Carlo test.

	Axis 1	Axis 2	Axis 3	Axis 4
Eigenvalue	0.408	0.256	0.240	0.177
Species - environment correlation	0.906	0.855	0.816	0.795
Cumulative percentage variance				
of species data	10.2	16.6	22.6	27.1
of species - environment relation	31.8	51.7	70.4	84.2
Sum of all eigenvalues	3.993			
Sum of all canonical eigenvalues	1.284			
Variance explained by CCA (%)	32.16			
	F	*p*		
Significance of first canonical axis	3.414	0.057		
Significance of all canonical axes	2.369	0.001		

CCA plots (Figure 4.3a and b) show phytoplankton species associated with the summer community plotting towards higher surface water temperatures and increased abundance of *Bosmina* spp. during late summer and autumn. Phytoplankton species representative of this trend include the cyanobacteria *Oscillatoria* spp. and *Microcystis* sp., and chlorophytes including the genera *Scenedesmus* and *Coelastrum*. Samples representing the early-summer clear-water phase plot towards the right-hand side of Figure 4.3b and are associated with increased abundances of *Daphnia* spp. and the phytoplankton genera *Ankrya* and *Chlorella*. A cluster of phytoplankton species on the right of Figure 4.3a appear to be associated with their presence in the phytoplankton community during June 2006.

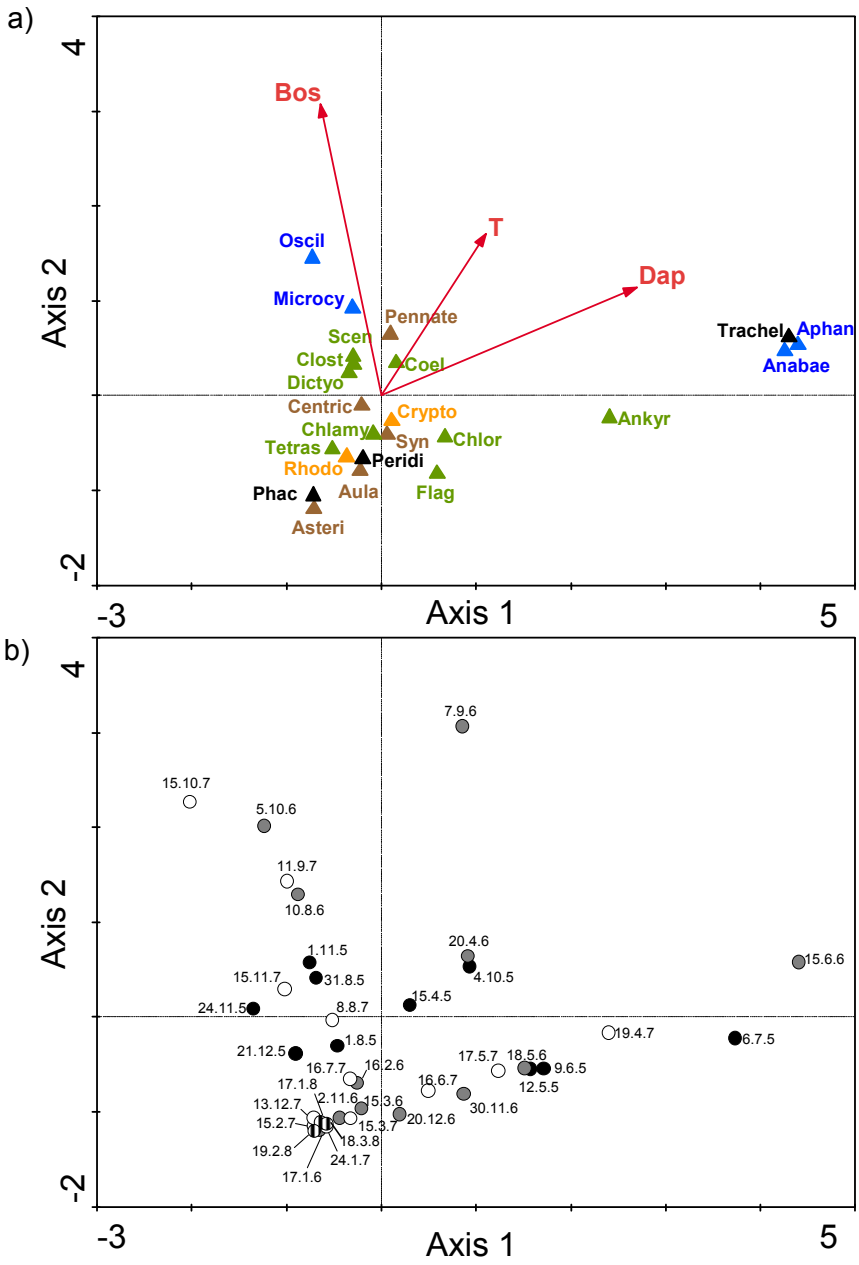

Figure 4.3 CCA plots of Lake C3 (Main Pond) constrained by the environmental variables. Only the first and second axes are shown. a) Phytoplankton species colour coded by phytoplankton group: brown, Bacillariophyceae; green, chlorophytes; orange, cryptophytes; blue, cyanobacteria; black, others (including chrysophytes and dinophytes). Phytoplankton abbreviations are given in Table 3.1, page 178 and environmental variable abbreviations in Table 4.1, page 200. b) samples, labelled by date (d.m.y) and colour-coded by year: black, 2005; grey, 2006; white, 2007; striped, 2008.

Wintertime samples are concentrated in the lower left-hand side of Figure 4.3b, and were mainly associated with diatoms, cryptophytes and euglenophytes. Phytoplankton species representative of this trend include the cyanobacteria *Oscillatoria* spp. and *Microcystis* spp., and chlorophytes including the genera *Scenedesmus* and *Coelastrum*. Samples representing the early-summer clear-water phase plot towards the right-hand side of Figure 4.3b and are associated with increased abundances of *Daphnia* spp. and the phytoplankton genera *Ankyra* and *Chlorella*. A cluster of phytoplankton species on the right of Figure 4.3a appear to be associated with their presence in the phytoplankton community during June 2006. Wintertime samples are concentrated in the lower left-hand side of Figure 4.3b, and were mainly associated with diatoms, cryptophytes and euglenophytes.

4.3.4 Lake I1 (Church Pond)

Only three variables that were positively correlated with chlorophyll-*a* concentrations were identified in Lake I1, and the three negatively correlated variables were the zooplankton groups calanoid copepods and *Ceriodaphnia* sp. and total alkalinity (Table 4.9). Neither the strongest positive or negative correlations were as strong as those in the connected lakes.

Table 4.9 Spearman's rank correlation coefficients between chlorophyll-*a* concentration and zooplankton and environmental variables in Lake I1 (Church Pond) March 2005-March 2008.

Variable	Spearman's rank correlation	
pH	0.435	
SiO_3	0.432	
Rotifers	0.400	
Calanoid		-0.543
Total alkalinity		-0.445
Ceriodaphnia		-0.365

CCA showed that eight environmental variables were independently and significantly correlated with phytoplankton community variance after forward selection. The phytoplankton genus *Dinobryon* was omitted from analysis as it was an outlier within the dataset. Only nutrients and physico-chemical parameters were significant in the CCA. Axis 1 explained 10.1% of phytoplankton community variance (Table 4.10) and

was most strongly correlated with NO_3-N concentrations (r = 0.651) and total alkalinity (r = 0.644). pH, temperature, SRP and SiO_3 concentrations were all negatively correlated with axis 1. The second axis was most strongly correlated with SRP and NH_4-N concentrations (r = -0.453 and -0.410 respectively) and explained an additional 7.6% of phytoplankton community variance. Together, axes 3 and 4 explained an additional 11.3% of phytoplankton community variance. Axis 3 was negatively correlated with pH (r = -0.501) and positively correlated with specific conductivity (r = 0.449). Axis 4 was most strongly negatively correlated with conductivity (-0.240) positively correlated with SiO_3 concentrations (r = 0.202).

Amongst the variables in the CCA, SRP was negatively correlated with conductivity (r = -0.551) and total alkalinity (r = -0.447). pH and total alkalinity were both strongly negatively correlated (r = -0.622). NO_3-N and NH_4-N were negatively correlated with temperature (r = -0.502 and -0.500 respectively) and pH was positively correlated with temperature (r = 0.592) NH_4-N and NO_3-N were positively correlated (r = 0.468).

Table 4.10 Summary of the results of CCA analysis between phytoplankton species and environmental variables and zooplankton abundance in Lake I1 (Church Pond), March 2005 to March 2008. Significance values are derived from a 1000 permutation Monte Carlo test. The outlying genus *Dinobryon* was removed from this analysis.

	Axis 1	Axis 2	Axis 3	Axis 4
Eigenvalue	0.430	0.322	0.254	0.225
Species - environment correlation	0.895	0.870	0.846	0.785
Cumulative percentage variance				
of species data	10.1	17.7	23.7	29.0
of species - environment relation	24.9	43.6	58.3	71.3
Sum of all eigenvalues	4.245			
Sum of all canonical eigenvalues	1.725			
Variance explained by CCA (%)	40.63			
	F	*p*		
Significance of first canonical axis	3.045	0.003		
Significance of all canonical axes	1.725	0.001		

CCA plots show that cyanobacteria were associated with increased pH and SiO_3 concentrations (Figure 4.4a), most likely as a reflection of the increased pH and SiO_3 concentrations occurring at the end of summer 2006 when cyanobacteria were most

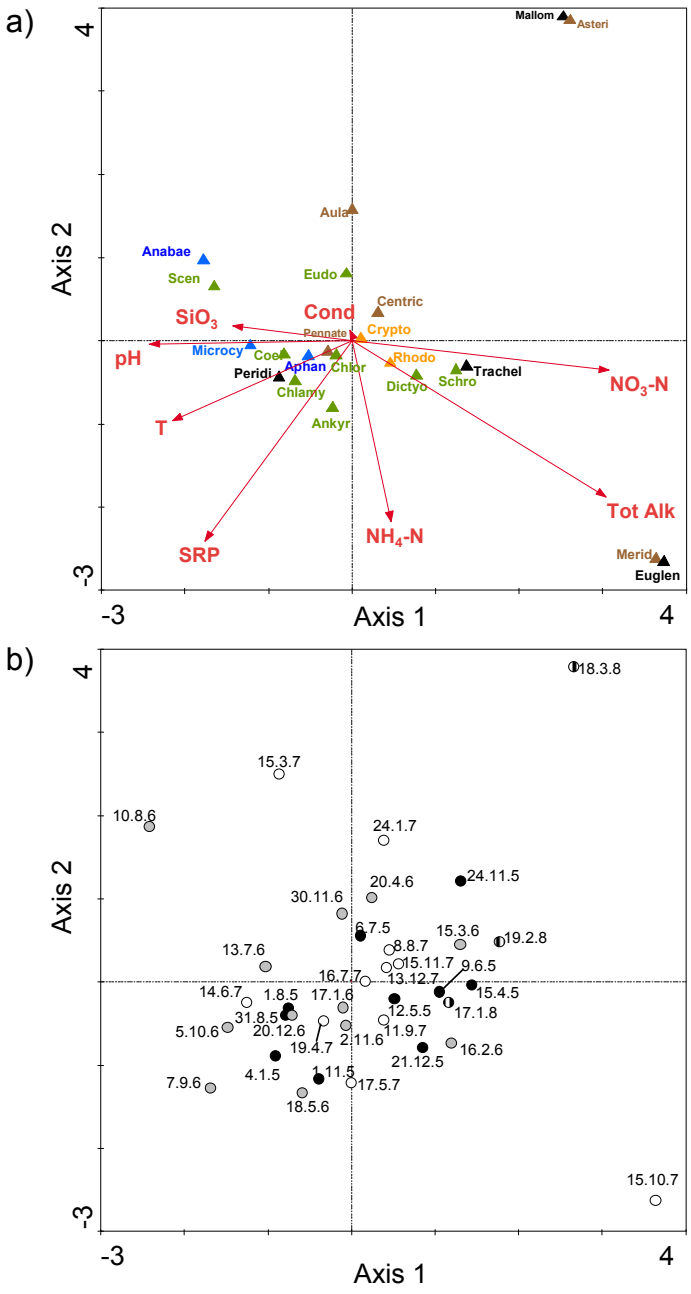

Figure 4.4 CCA plots of Lake I1 (Church Pond) constrained by the environmental variables. Only the first and second axes are shown. a) Phytoplankton species colour coded by phytoplankton group: brown, Bacillariophyceae; green, chlorophytes; orange, cryptophytes; blue, cyanobacteria; black, others (including chrysophytes and dinophytes). Phytoplankton abbreviations are given in Table 3.1, page 178 and environmental variable abbreviations in Table 4.1, page 200. b) samples, labelled by date (d.m.y) and colour-coded by year: black, 2005; grey, 2006; white, 2007; striped, 2008.

abundant. Samples from summer 2005 and 2006 plotted towards the left of Figure 4.4b due to the occurrence of cyanobacteria species during the summer. For example the August 2006 sample was associated with large biovolumes of *Anabaena* sp. Cyanobacteria were also positively associated with temperature and negatively related to NO_3-N concentrations. Summer 2007 samples occur closer to the centre and right of the plot reflecting the dominance of *Cryptomonas* spp. and pennate diatoms. Samples from November and December 2007 were also largely dominated by cryptophytes, and consequently plotted near the centre of Figure 4.4b. Negative association between temperature and cryptophytes and centric diatoms suggests their relative abundance during the winter. The March 2008 sample is strongly associated with *Asterionella*, which was rarely observed for the rest of the monitoring period.

4.3.5 Lake I2 (Clifton Pond)

Few correlations between chlorophyll-*a* and environmental variables were found in Lake I2 (Table 4.11). Only *Bosmina* spp. were positively correlated, albeit weakly, and similarly weak negative correlations were found for Mg^{2+} and *Daphnia* spp.

Table 4.11 Spearman's rank correlation coefficients between chlorophyll-*a* concentration and zooplankton and environmental variables in Lake I2 (Clifton Pond) March 2005-March 2008.

Variable	Spearman's rank correlation	
Bosmina spp.	0.340	
Mg+		-0.334
Daphnia spp.		-0.454

CCA analysis showed that 4 variables independently explained a significant proportion of variability in the phytoplankton community of Lake I2 after forward selection (Figure 4.5). Axis 1 was strongly positively correlated with NO_3-N concentrations (r = 0.654), and negatively correlated with pH (r = -0.448), and explained 6.3% of phytoplankton community variance (Table 4.12). 4% of variance in the phytoplankton community was explained by axis 2, which was most strongly correlated with K^+ (r = -0.631) and conductivity (r = -0.600). Axes 3 and 4 explained an additional 6.6% of phytoplankton community variability. Axis 3 was most strongly correlated with pH (r = -0.557) and axis 4 with NO_3-N concentration (r = 0.356). CCA indicated negative correlation

between NO_3-N and pH (r = -0.424). Weaker negative correlations also existed between NO_3-N and conductivity (r = -0.278) and K^+ (-0.268). Conductivity and pH were positively correlated (r = 0.309).

Cyanobacteria appeared to be associated with increased pH in Lake I2 (Figure 4.5a). Samples associated with cyanobacteria were mainly from summer 2005 and 2006. Summer 2007 samples were more closely associated with chlorophytes such as *Ankyra* spp., and also reflect the increased K^+ concentrations observed during the summer of 2007. The winter phytoplankton was related to *Pediastrum* spp., *Cocconeis* sp., *Euglena* sp. and *Trachelmonas* sp. Samples from January and March 2008 were closely associated with *Mallomonas* sp. and *Gymnodinium* sp. respectively. *Cryptomonas* spp. and *Rhodomonas* spp. plotted towards the centre of Figure 4.5b, due to their presence in most phytoplankton samples.

Table 4.12 Summary of the results of CCA analysis between phytoplankton species and environmental variables and zooplankton abundance in Lake I2 (Clifton Pond), March 2005 to March 2008. Significance values are derived from a 1000 permutation Monte Carlo test.

	Axis 1	Axis 2	Axis 3	Axis 4
Eigenvalue	0.331	0.208	0.194	0.152
Species - environment correlation	0.821	0.772	0.675	0.730
Cumulative percentage variance				
of species data	6.3	10.3	14.0	16.9
of species - environment relation	37.5	60.9	82.8	100.0
Sum of all eigenvalues	5.236			
Sum of all canonical eigenvalues	0.885			
Variance explained by CCA	16.90%			
	F	*p*		
Significance of first canonical axis	2.163	0.008		
Significance of all canonical axes	1.627	0.001		

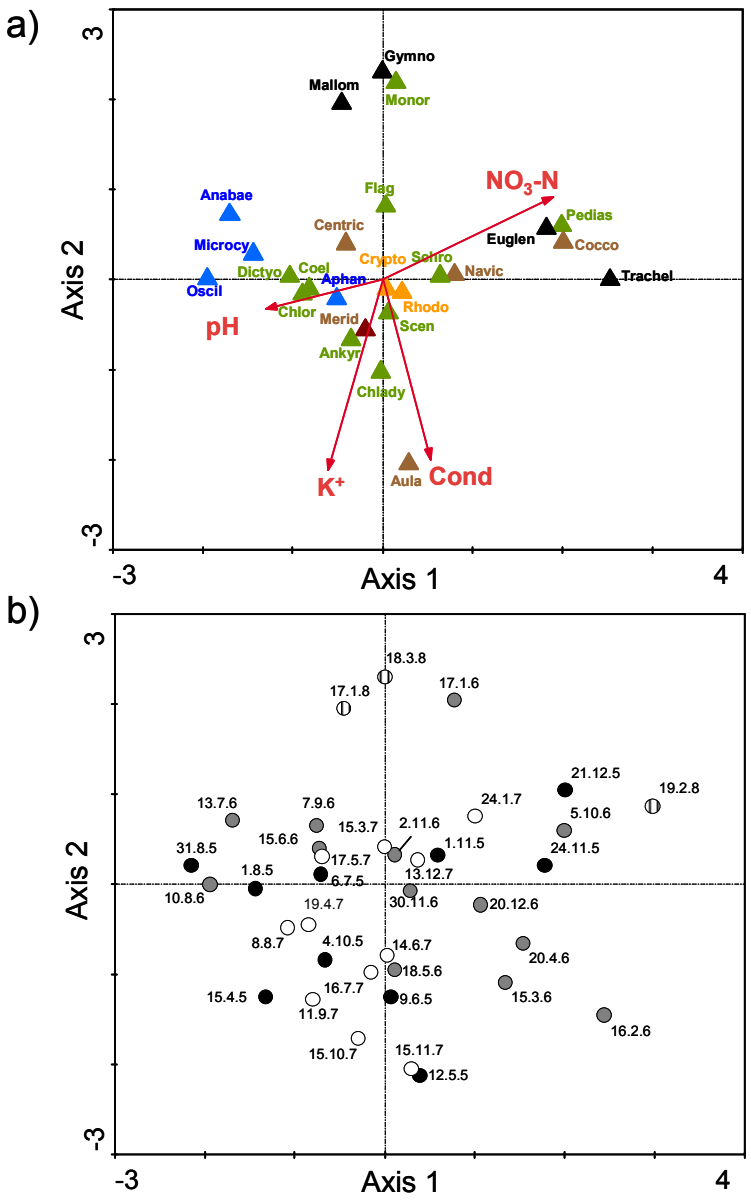

Figure 4.5 CCA plots of Lake I2 (Clifton Pond) constrained by the environmental variables. Only the first and second axes are shown. a) Phytoplankton species colour coded by phytoplankton group: brown, Bacillariophyceae; green, chlorophytes; orange, cryptophytes; blue, cyanobacteria; black, others (including chrysophytes and dinophytes). Phytoplankton abbreviations are given in Table 3.1, page 178 and environmental variable abbreviations in Table 4.1, page 200. b) samples, labelled by date (d.m.y) and colour-coded by year: black, 2005; grey, 2006; white, 2007; striped, 2008.

4.3.6 Lake I3 (Beeston Pond)

Only two variables were significantly correlated with chlorophyll-*a* concentrations in Lake I3 (Table 4.13). Strong negative correlation was found between NH_4-N and chlorophyll-*a*, and *Ceriodaphnia* spp. were weakly positively correlated with chlorophyll-*a* concentration.

Table 4.13 Spearman's rank correlation coefficients between chlorophyll-*a* concentration and zooplankton and environmental variables in Lake I3, (Beeston Pond) March 2005-March 2008.

Variable	Spearman's rank correlation
Ceriodaphnia sp.	0.414
NH_4-N	-0.725

A CCA of Lake I3 showed that two environmental variables independently and significantly explained phytoplankton community variance after forward selection. The first axis identified by CCA explained 6.4% of variance in the phytoplankton community (Table 4.14) and was strongly negatively correlated with SRP concentrations (r = -0.751). Axis 2, which explained an additional 4.4% of phytoplankton community variance, was strongly positively correlated with Na^+ concentrations. SRP and Na^+ concentrations were weakly negatively correlated (r = -0.329).

Table 4.14 Summary of the results of CCA analysis between phytoplankton species and environmental variables and zooplankton abundance in Lake I3 (Beeston Pond), March 2005 to March 2008. Significance values are derived from a 1000 permutation Monte Carlo test.

	Axis 1	Axis 2	Axis 3	Axis 4
Eigenvalue	0.260	0.177	0.562	0.428
Species - environment correlation	0.754	0.620	-	-
Cumulative percentage variance				
of species data	6.4	10.8	24.7	35.2
of species - environment relation	59.5	100.0	-	-
Sum of all eigenvalues	4.051			
Sum of all canonical eigenvalues	0.437			
Variance explained by CCA (%)	10.78			
	F	*p*		
Significance of first canonical axis	2.333	0.026		
Significance of all canonical axes	2.057	0.003		

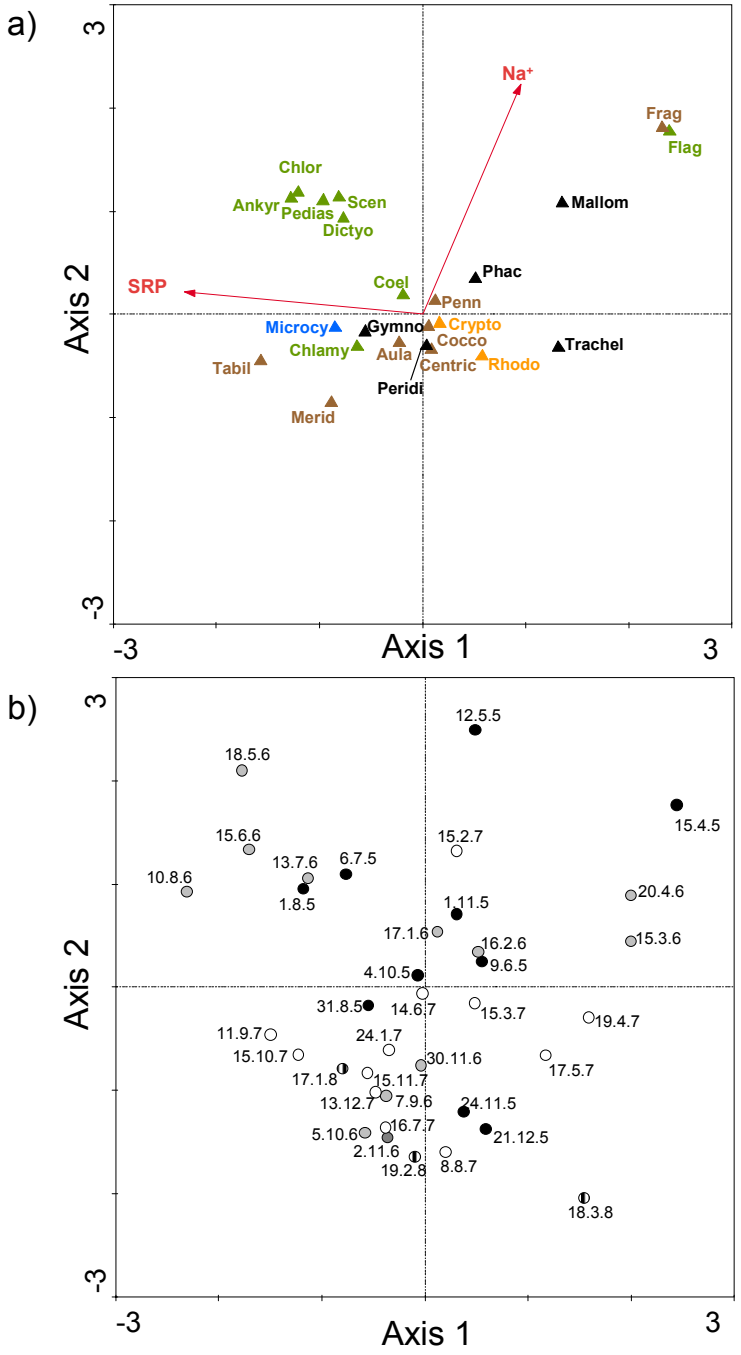

Figure 4.6 CCA plots of Lake I3 (Beeston Pond) constrained by the environmental variables. Only the first and second axes are shown. a) Phytoplankton species colour coded by phytoplankton group: brown, Bacillariophyceae; green, chlorophytes; orange, cryptophytes; blue, cyanobacteria; black, others (including chrysophytes and dinophytes). Phytoplankton abbreviations are given in Table 3.1, page 178 and environmental variable abbreviations in Table 4.1, page 200. b) samples, labelled by date (d.m.y) and colour-coded by year: black, 2005; grey, 2006; white, 2007; striped, 2008.

Figure 4.6a shows a close association of most chlorophytes found in Lake I3, such as *Scenedesmus* spp., *Pediastrum* spp. and *Ankyra* sp., representing their co-occurrence in the phytoplankton community. Samples from summer 2005 and 2006 were particularly associated with these chlorophyte species (Figure 4.6b). Summer samples from 2005 and 2006, and most chlorophyte species, were associated with increases in SRP concentrations observed during July and August each year. Summer 2007 samples tended to have lower Axis 2 scores than samples from summer 2005 and 2006, reflecting an increase in the abundance of diatoms during 2007. Autumn and winter samples generally had negative axis 1 and 2 scores, which corresponds to the reductions in Na^+ concentrations which were observed during the winter. *Cryptomonas* spp. plotted close to the centre of Figure 4.6a, reflecting the presence of *Cryptomonas* in the phytoplankton community throughout the monitoring period.

4.4 Discussion

Negative correlations between NH_4-N and chlorophyll-*a* concentrations suggest that NH_4-N strongly depleted by phytoplankton uptake in the connected lakes and Lake I3 (Table 4.15). NH_4-N was not significantly associated with chlorophyll-*a* concentrations in lakes I1 and I2. Negative NH_4-N correlations with chlorophyll-*a* concentrations have been observed by other authors in eutrophic shallow lakes (e.g. Lau and Lane, 2002). NH_4-N is the preferred source of N for phytoplankton (Wetzel, 1983; Brönmark and Hansson, 1998), and therefore NH_4-N concentrations are likely to be depleted when phytoplankton growth is rapid. Internal recycling of NH_4-N may be important in supplementing the supply of NH_4-N from the River Erewash during during the summer. Internally recycled NH_4-N was the most important source of N for phytoplankton in Lake Balaton, Hungary where rapid increases in phytoplankton biomass could lead to NH_4-N limitation during late summer (Présing et al., 2008). Rapid increases in NH_4-N concentration often occurred after the spring phytoplankton bloom and during autumn in the connected lakes of Attenborough Nature Reserve, suggesting that excretion and the sedimentary release are likely to be important for internal loading in these lakes. Sedimentary release of NH_4-N may be particularly important when inputs of organic matter to the sediments and temperatures are high (van Luijn et al., 1999). The release of NH_4-N from sediments may be increased under

anoxic conditions (Beutel, 2006) and hence support higher deposition of senescent phytoplankton, creating a positive feedback to enhance eutrophication (van Luijn et al., 1999).

Table 4.15 Summary of variables significantly ($p \leq 0.05$) correlated with chlorophyll-*a* concentration (Spearman's correlation) and associated with phytoplankton community structure (CCA). Abbreviations: Cyclop. cops., cyclopoid copepods; Calan. cops., calanoid copepods; Spec. cond., specific conductivity.

Lake	Spearman's correlations		CCA
	Positive	Negative	
C1	Total alkalinity Mg^{2+} Cyclop. cops. *Bosmina* spp. Temperature pH	NH_4-N NO_3-N SiO_3	NH_4-N NO_3-N SiO_3 *Daphnia* spp. Temperature Mg^{2+}
C2	TP Total alk K^+ Mg^{2+} Cyclop. cops. *Bosmina* spp. Temperature Spec. cond. pH	NH_4-N NO_3-N SiO_3	TP *Daphnia* spp. Calan. cops. Ceriodaphnia *Bosmina* spp. Ca^{2+}
C3	TP Total alkalinity Na^+ K^+ pH Temperature	NH_4-N NO_3-N SiO3	*Daphnia* spp. *Bosmina* spp. Temperature
I1	SiO_3 pH Rotifers	Total alkalinity *Ceriodaphnia* spp. Calan. cops.	NH_4-N NO_3-N SRP SiO_3 Total alkalinity Temperature pH Spec. cond.
I2	*Bosmina* spp.	Mg^{2+} *Daphnia* spp.	NO_3-N K^+ pH Spec. cond.
I3	*Ceriodaphnia* spp.	NH_4-N	SRP Na^+

Negative correlations between chlorophyll-*a* and NO_3-N concentrations in the connected lakes may reflect uptake of NO_3-N by phytoplankton. Although NH_4-N is the

preferred source of N for phytoplankton, Dortch (1990) suggests that NO_3-N may maintain phytoplankton growth rates in the absence of NH_4-N. The depletion of NH_4-N to undetectable levels during summer in the connected lakes (e.g. Figure 3.2b, page 94) suggests that this may occur. Uptake of NO_3-N by phytoplankton may have contributed towards a negative correlation with chlorophyll-*a* concentrations in the connected lakes (Table 4.15). NO_3-N denitrification may also have reduced water-column NO_3-N concentrations during the summer (Wetzel, 1983; Bennion and Smith, 2000). Bergin and Hamilton (2007) have argued that the reduction of NO_3-N to NH_4-N may also account for a significant loss of NO_3-N from aquatic systems. In isolated lakes I1 and I2, NO_3-N was a significantly related to phytoplankton community structure (Table 4.15). Uptake of NO_3-N by submerged macrophytes in these lakes may contribute towards low N concentrations during the summer. Submerged macrophytes can obtain nutrients from sediments as well as the water column and so are effective competitors for nutrients (Van Donk and Van de Bund, 2002). Species of phytoplankton present in both lakes I1 and I2 had heterocysts (e.g. *Aphanizomenon* sp. and *Anabaena* sp.) suggesting that N-limitation may affect the phytoplankton community.

SiO_3 concentrations were negatively associated with chlorophyll-*a* concentration in the connected lakes. Rapid decreases and increases in SiO_3 concentrations were observed in these lakes, probably because of the high biovolumes of planktonic diatoms in the phytoplankton of the connected lakes. High productivity of diatoms has been observed to be related to high uptake of Si (Bailey-Watts, 1976; Gibson et al., 2000). In the connected lakes, Si was frequently rapidly depleted to limiting levels (~0.5 mg L^{-1}, Reynolds, 2006) during spring and autumn which probably reflects the importance of diatoms in the phytoplankton community during those periods (see Chapter 3, Figure 3.5, Figure 3.13, Figure 3.21). It is likely that the uptake of Si by diatoms exhausted the Si available for uptake provided by the River Erewash, and therefore internal recycling may have been of considerable importance for supplementing the external supply (e.g. Parker et al., 1977; Gibson et al., 2000; Miretzky and Cirelli, 2004). Higher temperatures may increase the release of Si from sediments, suggesting that internal recycling could be greater during the spring and summer (Rippey, 1977, 1983).

Zooplankton appeared to exert more control over total phytoplankton biomass in lakes I1 and I2 than in the other lakes. Negative correlations between zooplankton (calanoid copepods, *Ceriodaphnia* and *Daphnia* spp.) and chlorophyll-*a* concentrations were found. This may be related to the abundance of submerged vegetation. The refuge hypothesis (Scheffer et al., 1993; Schriver et al., 1995; Stansfield et al., 1997; Perrow et al., 1999) suggests that submerged vegetation provides a refuge for zooplankton from zooplanktivorous fish, resulting in more large-bodied zooplankton which are normally susceptible to predation (Brooks and Dodson, 1965). The high grazing efficiency of larger zooplankton reduces the total abundance of phytoplankton, particularly of small centric diatoms and single-celled chlorophytes. This may contribute towards the proportional dominance of larger phytoplankton, such as cyanobacteria, that are grazing resistant (Bergquist et al., 1985) and may explain the relatively weak nature of these correlations. In the connected lakes, predation on zooplankton by zooplanktivorous fish is likely to be higher because of a lack of refuges. Therefore larger-bodied zooplankton are scarce (Brooks and Dodson, 1965), which reduces grazing pressure on phytoplankton and may favour the growth of smaller, rapidly reproducing phytoplankton species that can outcompete slower growing species for light and nutrients. Postive associations of zooplankton groups and chlorophyll-*a* concentrations were also found in all lakes except Lake C3 (Table 4.15). This probably reflects simultaneous seasonal increases in phytoplankton biomass and zooplankton abundance. The abundance of zooplankton groups tended to increase in the early summer and be most abundant during summer when chlorophyll-*a* concentrations were also highest.

However, CCA suggested that top-down control by zooplankton may have played a role in determining the phytoplankton community in the connected lakes. This may have been particularly strong during spring. Grazing by *Daphnia* spp. was probably the main cause of a spring clear-water phase. Samples from early summer in all of the connected lakes were associated with *Daphnia* spp. and a phytoplankton community that may be more easily grazed. Small centric diatoms and chlorophytes such as *Scenedesmus*, *Ankyra* and *Monoraphidium* were frequently found in the phytoplankton community during early summer in the connected lakes (e.g.Figure 3.14, page 114). These species are all susceptible to population losses by filter-feeding zooplankton (Agrawal, 1998; Reynolds, 2006). In eutrophic lakes, strong grazing control and the

creation of a clear-water period during the spring is a well known phenomenon (e.g. Vanni and Temte, 1990; Talling, 2003; Lampert and Sommer, 2007). The higher flushing rate of the connected lakes may favour the growth of small, fast reproducing phytoplankton (e.g. Brook and Woodward, 1956; García de Emiliani, 1997; Huszar and Reynolds, 1997; Paidere et al., 2007) which would reinforce the strength of top-down control of the phytoplankton. Timms and Moss (1984) related strong grazing control by *Daphnia* and a lack of cyanobacteria to a hydrological regime which promoted smaller phytoplankton species. In contrast, clear-water periods during spring were less well defined in lakes I1 and I2, mainly because the chlorophyll-*a* concentrations did not increase during the summer as much as those in the connected lakes did.

Some significant correlations between chlorophyll-*a* and the environmental variables are probably not directly related, but may reflect other aspects, such as seasonality. Major ion and chlorophyll-*a* concentrations were strongly correlated in the connected lakes (Table 4.15). For example, in lakes C2 and C3, significant positive correlation between K^+ and chlorophyll-*a* reflects the low concentrations of both variables during the winter months (see Figure 3.10, page 120 and Figure 3.18, 120). Major ions are not strongy mediated by phytoplankton uptake (e.g. Krivtsov and Sigee, 2005; Reynolds, 2006) which suggests that the correlations observed in this analysis are likely to be indirectly related by seasonal variations in hydrology, such as dilution during the winter and evaporative concentration during summer.

It is possible that the sampling technique deployed led to some errors in the estimates of zooplankton biomass. The data used here was obtained from tows at a single pelagic sampling site. In turbid lakes with little submerged vegetation, littoral macrophytes may be an important refuge for zooplankton (Cazzanelli et al., 2008) although complex interactions between fish predation, macrophyte avoidance and horizontal migration are likely to moderate this effect (Burks et al., 2001; 2002; Geraldes and Boavida, 2004). In isolated lakes I1 and I2 where submerged vegetation was abundant, swarms of large-bodied *Daphnia* were anecdotally observed amongst submerged macrophyte beds during summer, a behaviour observed elsewhere in the zooplankton (e.g. Colebrook, 1960; Jakobsen and Johnsen, 1988) and possibly related to predator avoidance (Walls et al., 1990). The analysis of pelagic samples

may therefore have underestimated total zooplankton biomass in both isolated and connected lakes, and therefore over-estimated the strength of bottom-up control of the phytoplankton.

4.5 Conclusion

The analysis presented in this chapter has demonstrated that the association of phytoplankton biomass and community structure with nutrients and zooplankton varies amongst the lakes of Attenborough Nature Reserve. Grazing by zooplankton has been identified as a potentially important mechanism for reducing phytoplankton biomass in the connected lakes during spring. The dominance of small phytoplankton species susceptible to grazing by zooplankton results in strong top-down control of phytoplankton biomass and the development of a clear-water phase. This is probably strengthened by the hydrological regime which favours the growth of smaller and disturbance-tolerant phytoplankton species.

Uptake of NO_3-N by phytoplankton probably occurred when NH_4-N was depleted, which in addition to denitrification during the summer is likely to explain the strong negative correlations between NO_3-N and chlorophyll-*a* concentration. N was probably more limiting to most non-heterocystous phytoplankton in lakes I1 and I2 than in the connected lakes.

In isolated lakes, negative association between total phytoplankton biomass and zooplankton groups suggests greater top-down control of phytoplankton compared to the connected lakes. In the connected lakes, grazing strongly reduced phytoplankton biomass during the spring but not during the summer. The sampling technique used to measure zooplankton may underestimate their abundance, particularly in the lakes with abundant submerged vegetation where zooplankton appeared to be more abundant.

Chapter 5 Effects of flood events on the lakes of Attenborough Nature Reserve

5.1 Introduction

The water balance of a lake is governed by the relative inputs and losses. This can be altered by catchment scale processes and climate variability (Wetzel, 1983). Additionally, specific flood events can cause substantial changes in the hydrology of a lake (e.g. Barbiero et al., 1999). At Attenborough Nature Reserve, hydrology may play an important role in helping to explain the differences in chemistry and biology observed during 2007 (Chapter 3). Future reductions in discharge into the connected lakes as a result of the River Erewash diversion may also alter the chemistry and biology of the connected lakes. Therefore it is important to determine how hydrological changes may affect the lakes of Attenborough Nature Reserve.

Distinct differences in lake depth and flushing rate were observed between 2005, 2006 and 2007 in both connected and isolated lakes. Examining the differences in chemistry and biology observed during these hydrologically different periods allows for the short-term effects of flooding in the Attenborough Ponds to be evaluated. This is of importance to shallow lake management given the potential for changes in discharge to lakes, such as from increased storminess related to climate change (IPCC, 2008) and greater flashiness in stream discharge as a result of urbanisation (Davie, 2002).

Diverting the River Erewash to flow directly into the River Trent will reduce the inflow from the River Erewash into the connected lakes. Hydrological modelling of the diversion (HR Wallingford, 2006) suggests the average annual inflow from the River Erewash to the connected lakes will be reduced to 6.25% of the current volume. The discharge of summer (1 May-1 October) storms (when discharge is in excess of 1.5 m^3 sec^{-1}) will be reduced to 6.67% of the current volume (HR Wallingford, 2006). Predicting how this reduction in flow will affect the chemistry and biology is important in developing future successful management plans for Attenborough Nature Reserve.

In order to address the effect of flood events and the possible response of connected lakes to reductions in discharge, the specific aims of this chapter are to:

1. Describe the changes in hydrological regime during the monitoring period,
2. Assess the importance of rivers and streams flowing into Attenborough Nature Reserve as suppliers of nutrients;
3. Investigate how hydrological change affects chemical and biological components of connected and isolated lakes;
4. Understand how future hydrological change (i.e. imminent reduction of River Erewash inflow to the connected lakes, and possible future enhanced flooding due to climate change) may affect the structure and functioning of connected lakes;
5. Recommend an optimal flushing regime for long-term recovery.

5.2 Methods

5.2.1 Connected lakes

Hydrology

River Erewash discharge data were obtained in order to assess the importance of river inflow for the water and nutrient budgets of the connected lakes. Discharge data were obtained from a Flowline Ultraflux ultrasonic gauging system installed and operated at site Eb (see Figure 2.1, page 74) by Cemex Operations U.K. Ltd. Flow was measured at 15 minute intervals from which data at 30 minute intervals were extracted. These were summed to estimate total discharge for the periods between sampling dates. Periods of missing data exist in the discharge record at Eb, associated with periods following high flows when debris blocked the flow meter. Since strong linear correlation exist between the mean daily discharge observed in the River Erewash and daily discharge in River Trent (sourced from the Environment Agency gauging station at Colwick, Nottinghamshire, $r^2 = 0.67$, $p \leq 0.001$), missing data were calculated daily using the regression equation:

$$R.Erewash = (0.023 \times R.Trent) + 0.20$$

Measurements of discharge from the connected lakes were made at sites Ea (a stream from Lake C1, discharging into the River Trent through a single flap) and MPO (a six-flap weir located along the north-eastern edge of Lake C3, see Figure 2.1, page 74, for locations) at approximately four-weekly intervals, usually on the day preceding lake sampling. Disharge measurements prior to October 2005 were made exclusively by S. McGowan, and from November 2005 to March 2008 measurements were obtained collaboratively by S. McGowan and the author. At site Ea, discharge was calculated by dividing the width of the stream into thirds. In each third of the stream width, the surface area of the stream was calculated by multiplying the water depth by the width of the section. One measurement of water velocity was made at approximately one-third of the water depth from the surface in each third, using a Valeport 801 electromagnetic flow meter to measure mean flow over a 30 sec period. Discharge was calculated by multiplying the area of each third by the discharge, and then summed together to give one estimate of discharge for Ea. The mean discharge for the whole monitoring period was derived using all samples. At site MPO, measurements were made in each of the six divisions in the weir (see Figure 5.9a). Discharge (m^3 sec^{-1}) was calculated by multiplying the velocity of the water in the weir division, estimated using a surface float, by the cross-sectional area of water in the weir (depth × width) (Davie, 2002). Calculations from each weir division were summed to form one estimate of discharge for the outflow. All discharge calculations at MPO were used to estimate mean discharge for the monitoring period.

The water retention time (WRT) of the connected lakes was calculated using estimated lake volume and the discharge data recorded at Ea, using interpolated data where required, on a daily basis. The volume of the connected lakes was based on depth measurements taken by S. McGowan during vegetation surveys in 2005 and the lake extent derived from GPS co-ordinates of littoral sites, and computed using Surfer 8.0 using point kriging to build a DEM at a 20 m resolution. Approximately 470 measurements of depth were included in the model. Bathymetric maps were presented in Figure 2.2 (page 76).

Nutrient budget

The concentrations of nutrients and major ions observed at each of the inflows and discharges were compared to the in-lake concentrations using the Wilcoxon signed-rank test computed using SPSS 15.0. The Wilcoxon signed-rank test *z*-score statistic is the standardised distance between the smallest sum of ranks (either positive or negative) and the expected rank (defined as the sum of ranks × 0.5). Regardless of the direction of the difference, the *z*-score is always negative. Shading is used in the tables of results to interpret the direction of the difference and therefore determine if nutrient concentrations were higher in the lake or stream. Nutrient and major ion concentrations were also determined on River Erewash samples taken upstream of the Toton seweage treatment works (STW) discharge (site Ebb, Figure 2.1, page 74) from July 2006, in order to compare with nutrient concentrations downstream of the STW discharge (site Eb).

A simple mass-balance nutrient budget for the connected lakes was developed in order to gain an insight into the dynamics of nutrient storage and release, and to investigate the relevance of internal processes in determining in-lake nutrient concentrations. Monthly loading rates to the lakes were estimated by multiplying inflow (site Eb) and outflow (sites Ea and MPO) discharge volumes (m^3) by the measured concentration of nutrient (µg L^{-1}) (e.g. Beklioglu et al., 1999). The difference between the estimated quantity of nutrients entering at inflows and leaving at outflows was assumed to represent the retention of the nutrient within the lakes. To gain a further insight into TP dynamics, a Vollenweider-type model was developed, using the equation

$$TP_{(lake)} = L/z(\rho+\sigma)$$

where L was TP loading (mg m^3 $year^{-1}$); z, mean depth; ρ, lake flushing rate (defined as Q/V_L, where Q is inflow volume and V_L is lake volume), and σ is a coefficient for in-lake TP loss (Brett and Benjamin, 2008). L (TP loading) was calculated by dividing the total input of TP (discharge × concentration) by the estimated lake volume. Mean depth was estimated based on measurements of depth taken during vegetation surveys in 2005. Since no direct measurement of σ was available, an estimate was made using the equation $\sigma = \sqrt{(z/Q)} / t_q$ where t_q is hydraulic residence time (years)

and defined as V_L/Q (Vollenweider 1976, in Reynolds, 2006). Chlorophyll-*a* estimates were made based on the mean annual in-lake TP concentration using the equation log[chl-a] = 0.91[logTP] –0.435 (Vollenwieder and Kerekes, 1980, in Reynolds, 2006). The relationship between nutrient loading to the connected lakes and monthly discharge of the River Erewash was assessed using Spearman's rank correlation coefficient.

5.2.2 Isolated lakes

Water depth in lakes I1 and I2 was estimated by measuring against fixed points in order to assess the depth response of isolated lakes to flood events. In Lake I1 this was a platform built in the centre of the lake for bird nesting (see Figure Figure 3.25, page 133), and in Lake I2 depth measurements were made against a fixed fence post (see Figure 3.33, page 146). Data for Lake I1 represented the depth below a fixed point, so in order to facilitate comparison with Lake I2, values were multiplied by -1. For both lakes I1 and I2, depth measurements were converted to *z*-scores to allow for a comparison of variance. *z*-scores represent the number of standard deviations a measurement is from the mean, and were calculated by subtracting the mean from the original value and dividing by the standard deviation (Legendre and Legendre, 1998). Supplementary data of Lake I2 water depth (for the years 1995, 2000-2004, and 2007-2008), obtained from Sandy Aitken (pers. comm.) were derived by measuring water depth at a fixed point on the lake shore and converted to meters above sea level (m. A.S.L:). No water depth measurements were made in Lake I3.

5.2.3 Regional meteorology and hydrology

Daily rainfall data (mm day^{-1}) were obtained from a Met Office weather station situated on the University of Nottingham's Sutton Bonington Campus (SK 505262), and summed to provide monthly totals. Flow data for the River Trent was obtained from the Colwick gauging station, located approximately 15 km east (downstream) of the Attenborough Ponds (SK 620399) and operated by the Environment Agency. The record extended from 1958 to 2007. Yearly mean discharges were calculated based on calendar years.

5.2.4 Isotopes

Water for isotope analysis was collected from all connected lakes and isolated lakes I1 and I2 in order to compare the relative importance of evaporation and river inflow for the lakes of Attenborough Nature Reserve. Samples were collected on the same day as water for chemical and biological analysis in airtight polyethylene bottles by either S. McGowan or the author, and were kept refrigerated at 4°C. Unfiltered samples of water were equilibrated with CO_2, using an IsoPrep 18 device for oxygen isotope analysis and mass spectrometry, performed on a SIRA (both VG IsoGas, Middlewhich, U.K.). Analysis was performed at the NERC Isotope Geosciences Laboratory, in Keyworth, Nottinghamshire. Isotopic ratios ($^{18}O/^{16}O$) and $\delta^{18}O$ (‰, parts per mille) are defined in relation to the international standard Vienna Standard Mean Ocean Water (VSMOW) by comparison with laboratory standards calibrated using NBS standards. Analytical precision is typically ±0.05‰ for $\delta^{18}O$ (Melanie Leng and Matthew Jones, pers. comm.). The isotope datasets presented in this thesis were provided by Melanie Leng and Matthew Jones.

5.2.5 Chemistry and biology

The methods by which water chemistry and biological data used in this chapter were obtained are described in detail in Chapter 2. The data presented in section 5.6 presents the data discussed in Chapter 3 on a year-by-year basis to draw comparisons between dry years and flood years. In order to assess changes in the phytoplankton and zooplankton community as a result of changes in hydrological regime, the CA scores derived in Chapter 3 (section 3.8, page 175) were used and plotted as a timeseries. The methods used for deriving CA scores are given on page 92. One CA was used to derive scores for all phytoplankton samples and one for all zooplankton samples.

5.3 Results - Hydrology

5.3.1 Rainfall

Daily rainfall recorded at Sutton Bonington in Nottinghamshire is shown in Figure 5.1. During 2005, rainfall was greatest during the autumn which was followed by a dry winter with few rainfall events. February and late July 2006 was a dry period, particularly during June and July when rainfall did not exceed 3 mm day^{-1}. During the autumn and winter of 2006, more rainfall events of a smaller magnitude occurred than during the same period in 2005. March and April 2007 were a relatively dry period. However, throughout the summer of 2007 rainfall events became more prolonged than summer 2005 and 2006, particularly from May to July. Despite a reduction in rainfall intensity during the autumn of 2007, several high intensity and prolonged rainfall events were recorded between late November 2007 and February 2008. Mean daily rainfall during the monitoring period was 1.47 (±0.18) mm in 2005, 1.74 (±0.17) mm in 2006 and 2.08 (±0.22) mm during 2007. The four highest monthly total rainfalls were recorded during 2007 (Figure 5.2).

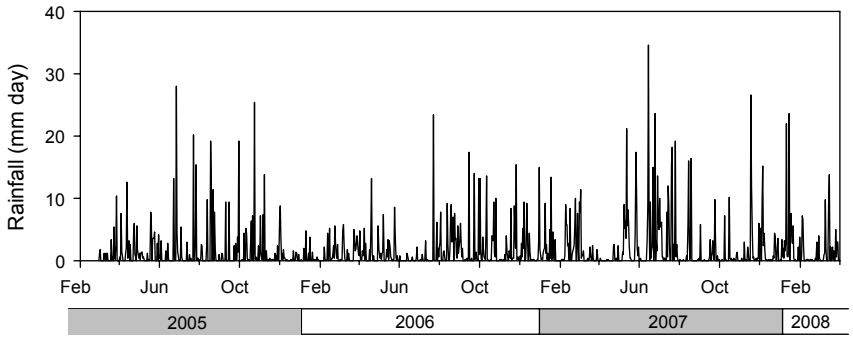

Figure 5.1 Daily rainfall recorded at Sutton Bonington, Nottinghamshire, March 2005 to March 2008.

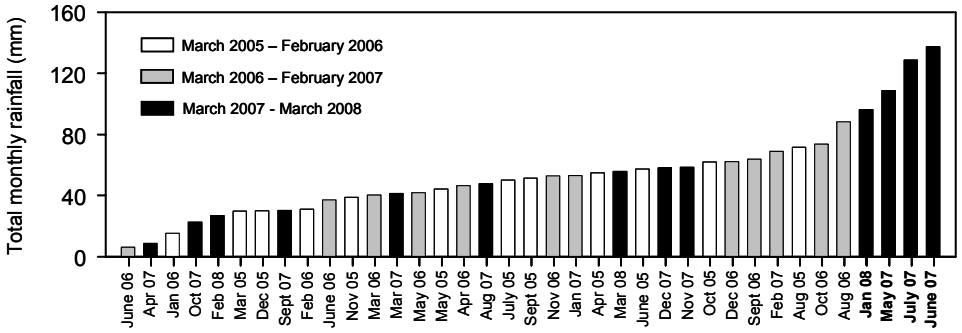

Figure 5.2 Total monthly rainfall recorded at Sutton Bonington, Nottinghamshire, March 2005 to March 2008.

5.3.2 River discharge

River Trent (long-term context)

Data for the River Trent suggest that discharge during 2005 and 2006 was low compared to several previous years (Figure 5.3a). The highest daily discharges recorded in the River Trent during the monitoring period generally occurred during October - January each year (Figure 5.3b). During winter 2006, discharge remained high until May 2007. Discharge in the River Trent during the summer of 2007 was considerably higher than during 2005 and 2006. The greatest discharge recorded during the monitoring period occurred during January 2008. The mean annual (March - March) discharges, calculated from daily values, were 62.1 (±2.1) and 70.1 (±2.4) m^3 sec^{-1} during 2005 and 2006 respectively, and were the 7^{th} and 13^{th} lowest mean discharges recorded between 1959 and 2006 (Figure 5.3c). In contrast, the mean discharge between March 2007 to March 2008 was the 4^{th} highest in the record (104.0 ±4.9 m^3 sec^{-1}), reflecting the high precipitation during summer and winter 2007 (Figure 5.2).

River Erewash

Discharge from the River Erewash at Eb was greater during 2007 than 2005 or 2006. During the monitoring period, River Erewash discharge ranged between <1 and 35 m^3 sec^{-1} (Figure 5.4). Discharge in the River Erewash rarely exceeded 10 m^3 sec^{-1} from March 2005 until January 2007 and was characterised by increases in discharge between October 2005 and February 2006. Other than a brief increase in discharge during May 2006, discharges during summer 2006 were generally lower than 2005. Discharge during January - March 2007 was higher than in 2006. This was followed by substantially greater discharges during June, July and August 2007 than the same period in previous years corresponding to the heavy precipitation observed during summer 2007 (Figure 5.2). Discharge was low during the autumn and winter of 2007, although discharge increased during January and February 2006.

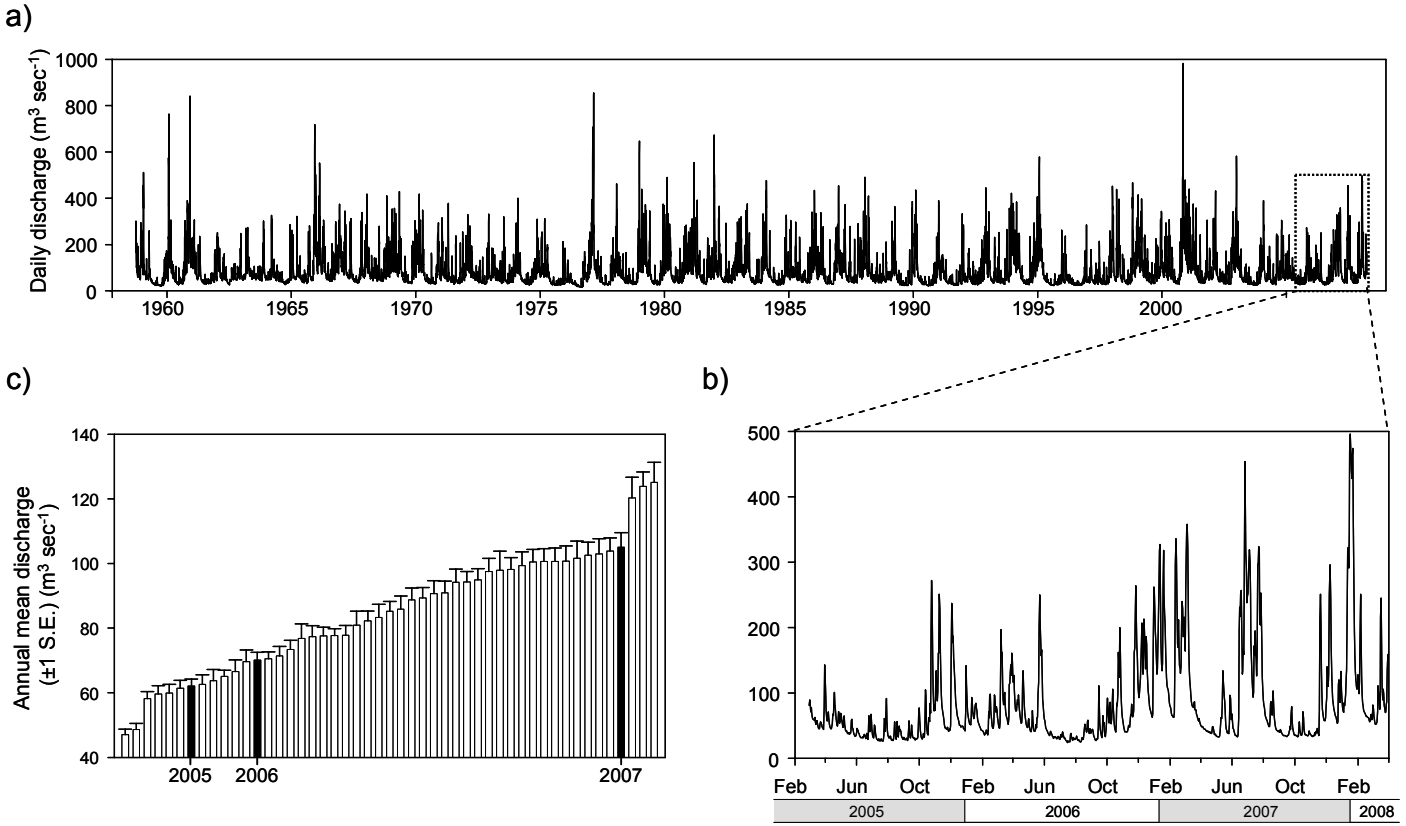

Figure 5.3 Discharge data for the River Trent obtained from the Colwick gauging station operated by the Environment Agency a) Daily discharge data (m^3 sec^{-1}), October 1958 to March 2008. b) Daily discharge data March 2005 to December 2006. c) Annual mean discharge data (+1 S.E.) highlighting 2005, 2006 and 2007.

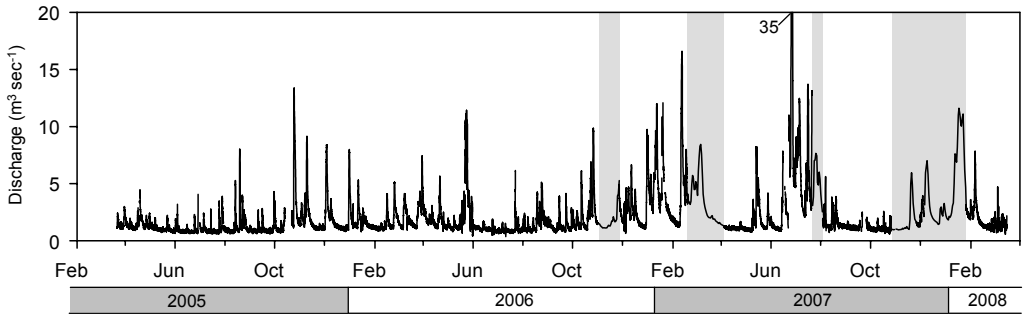

Figure 5.4 Discharge data for the River Erewash (gauged at site Eb, see Figure 2.1, page 74) at 30 min intervals, March 2005 to March 2008., courtesy of Cemex U.K. Operations Ltd. Shaded panels represent significant periods of missing data from Eb gauge, are mean daily discharges derived from interpolated River Trent daily discharge data.

5.3.3 Lake outflows

Ea (Lake C1 outflow to River Trent in the south of Attenborough Nature Reserve)

Water depth at Ea (see Figure 2.1, page 74) was highest during 2007 compared to 2005 and 2006. Water depth, measured approximately every four weeks, ranged between 0.56 and 1.80 m during the monitoring period (Figure 5.5). The mean was 1.00 (±0.05) m. Water depth was generally between 0.5 and 1.0 m during 2005, and was highest during August and November. Water depth decreased during 2006 and then rapidly increase between September 2006 and February 2007. Water depth remained above 0.8 m during 2007, unlike previous years, and exceeded 1.5 m in February and July. The greatest water depths during the monitoring period occurred in January and March 2008.

The mean discharge recorded at Ea during the monitoring period, calculated on a four-weekly basis, was 1.62 (±0.12) m^3 sec^{-1} and ranged between 0.02 and 3.11 m^3 sec^{-1} (Figure 5.5). Discharge at Ea was observed throughout 2005 and 2006, and was generally lowest during the summer of each year. During 2007, stagnation of discharge was observed during February and again during winter 2007 and early 2008, and the highest discharge occurred during July.

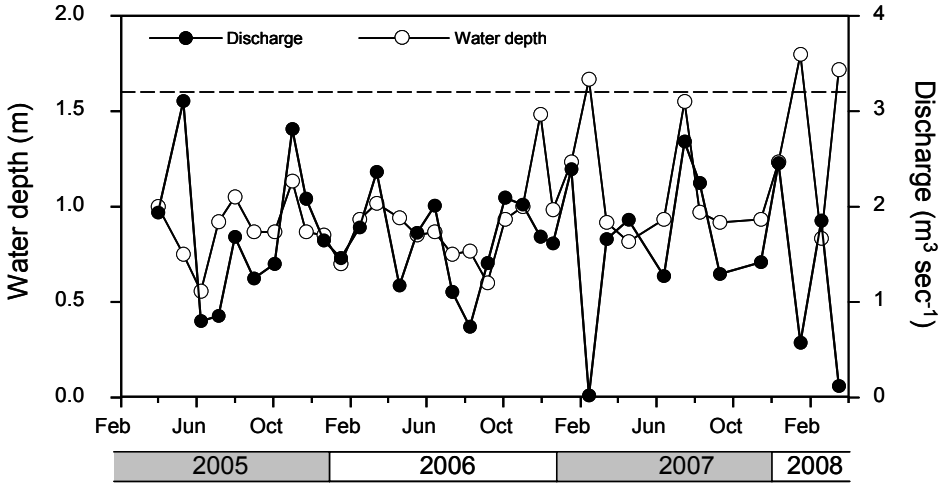

Figure 5.5 Water depth and estimated discharge at Ea, March 2005 to March 2008. The dashed line indicates an inferred threshold when discharge no longer increases with water depth.

Water depth and discharge were positively correlated (r_s = 0.483, p = 0.004) below a threshold of 1.6 m (Figure 5.6) at the Ea outflow. Heavy rainfall events resulted in increases in water depth in the outflowing stream and in the River Trent. When depths at Ea exceeded 1.6m, the positive linear correlation between water depth and discharge broke down because of stagnation of the outflow stream (Figure 5.7). Stagnation was associated with increases in the depth of the River Trent to above the height of the Ea discharge. Discharge at Ea was reduced to less than 0.5 m^3 sec^{-1} when the water depth at Ea was greater than 1.6m.

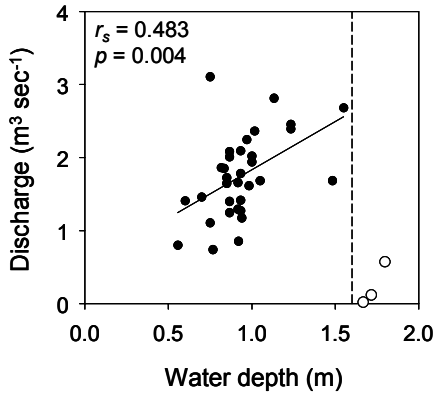

Figure 5.6 Water depth and estimated discharge at Ea, March 2005 to March 2008. The dotted line indicates the 1.6m depth threshold (see Figure 5.5).

Figure 5.7 Outflow at Ea showing stagnation of flow during wet periods. The River Trent is on the left of the photograph, taken in January 2008.

MPO (Lake C3 outflow to River Trent, along north-eastern boundary of Attenborough Nature Reserve)

Water depth at MPO ranged between 0 and 0.50 m (Figure 5.8). Mean water depth during the monitoring period, based on measurements made on approximately four-weekly intervals, was 0.06 (±0.02) m. Water did not exit at the MPO outflow during 2005 until December when the water depth reached 0.05 m. Water depth increased slightly during spring 2006, and for the majority of summer 2006 no water exited at MPO. Throughout 2007, water depth was greater than in previous years. Water depth was briefly high during February 2007 (0.5 m). Water depth increased from 0.02 to 0.10 m during early summer and increased sharply during July 2007. Water depth was low throughout autumn and winter 2007, but increased and remained over 0.2 m between January and March 2008.

The mean discharge at MPO (Figure 2.1, page 74) was 2.67 (±1.07) m^3 sec^{-1} during the monitoring period and ranged between 0 and 26.28 m^3 sec^{-1}. Discharge at MPO did not occur substantially during 2005 (Figure 5.8). Low discharges occurred during October 2005 to May 2006 and did not exceed 0.5 m^3 sec^{-1} during that period. No discharge occurred during the summer of 2006 until September 2006 after which discharge increased to 7.5 m^3 sec^{-1} in February 2007. Comparatively high discharges were recorded during the spring of 2007, particularly in May (9.0 m^3 sec^{-1}). Three notable events of high discharge were observed during the remainder of the monitoring period. Discharge exceeded 20 m^3 sec^{-1} in July 2007, January 2008 and March 2008.

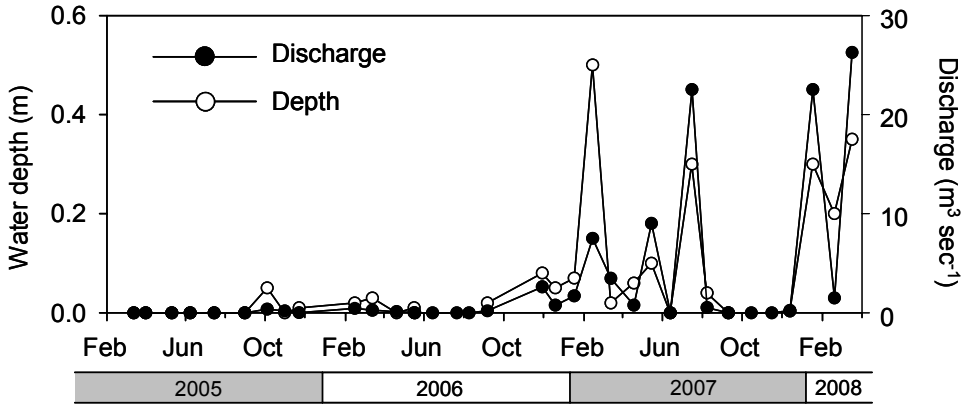

Figure 5.8 Estimated discharge and depth measurements at MPO, March 2005 to March 2008.

a)

b)

Figure 5.9 Outflow from connected lakes to the River Trent from the six-flap weir at MPO during a, wet period (January 2008), and b, dry period (August 2006). The River Trent is on the left of the photographs.

5.3.4 Connected lake hydrology

Flow patterns through connected lakes

Varying the discharge into the Erewash-connected lake chain altered the flow route of water. Figure 5.10 shows the relative discharges at Ea and MPO outflows during the monitoring period, and schematic maps of flow routes in each year are shown in Figure 5.11. During 2005, all outflows from connected lakes left through Ea until December, when 18% was observed flowing through MPO. Until June 2006, no more than 20% of outflow occurred through MPO, and by July 2006 all outflow passed through Ea. During 2007, MPO accounted for a higher proportion of outflow than the previous two years. From November 2006 to May 2007, over 30% of outflow left through MPO, reaching a maximum of 99% in February 2007. Ea became the dominant outflow during the early summer of 2007, although in July MPO again became the dominant outflow. Outflow was principally through Ea throughout the remainder of 2007, and then MPO during January to March 2008.

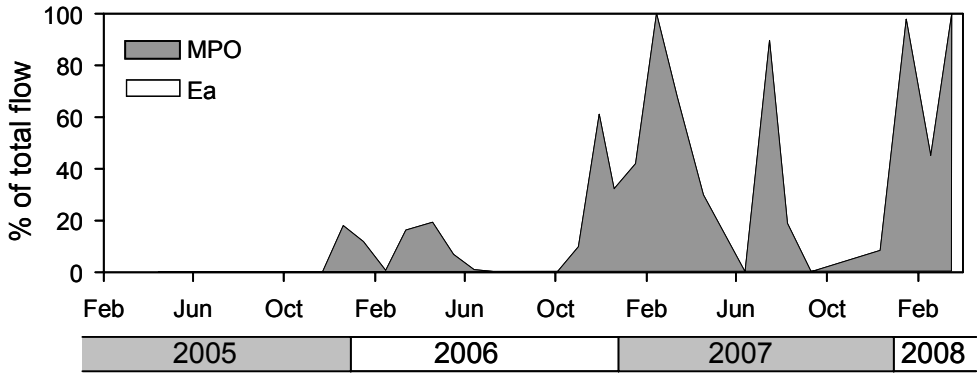

Figure 5.10 Percentage of total discharges observed at outflow sites MPO and Ea, March 2005 - March 2008.

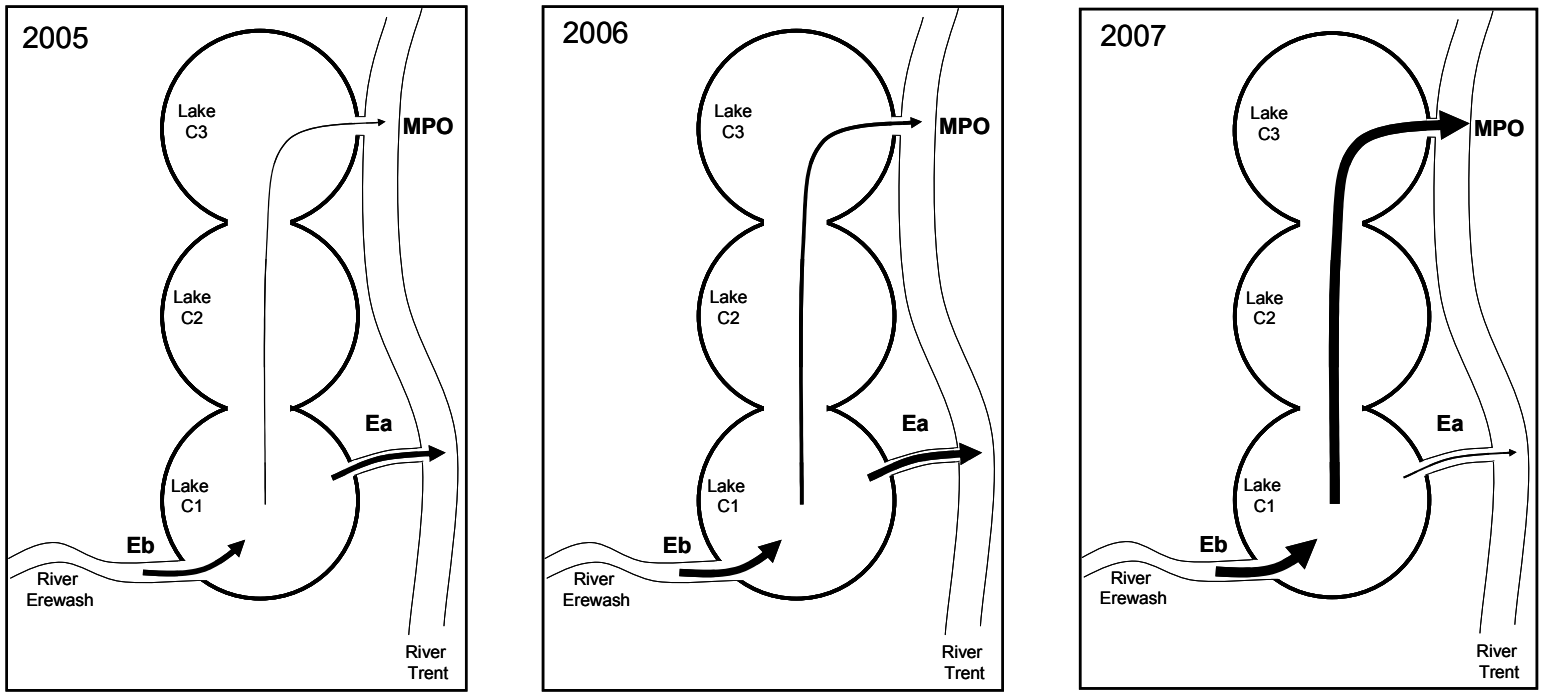

Figure 5.11 Schematic maps to show the relative flows of water into and out of connected lakes of Attenborough Nature Reserve during the years of the monitoring period. Thicker lines denote greater importance of flow during each year.

Water Retention Time

Daily water retention time for connected lakes ranged between 1.0 and 25.4 days and the mean was 13.8 (±0.2) days during the monitoring period (Figure 5.12). The WRTs observed during 2007 were frequently lower compared to 2005 and 2006. During both 2005 and 2006, water retention times increased during the spring and tended to be greatest during the summer. During the winters of 2005 and 2006, WRTs were generally between 7 and 15 days except for occasional decreases to ~3 days. In 2007, WRTs were lower during February than 2005 or 2006 and during in June and July 2007, WRTs were exceptionally low, approaching 2 days between mid-June and the end of July. Prolonged periods of low WRT were also recorded during the winter 2007 and January 2008.

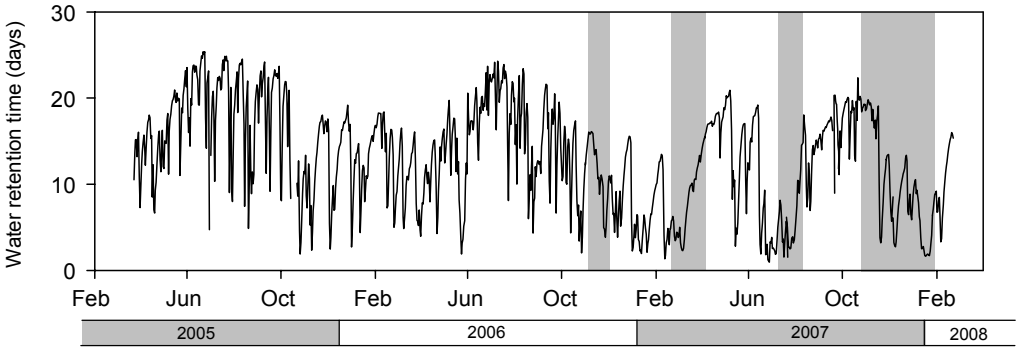

Figure 5.12 Estimated daily water retention time for connected lakes, March 2005 - March 2008. Shaded panels correspond to periods when daily discharge at Eb was estimated by regression using River Trent discharge (see section 5.2.1 for details).

Isotopic variability

Stable oxygen isotope ratios clearly distinguished between connected and isolated lakes throughout the monitoring period (Figure 5.13 and Figure 5.14). Because the lighter oxygen isotope ^{16}O is evaporated preferentially over the heavier ^{18}O, oxygen isotopes are increasing light with increasing evaporation. Lakes with longer residence times are therefore isotopically heavier (less depleted in ^{18}O) (Darling et al., 2005). Isolated lakes plotted further from the Global Meteoric Water Line (GMWL) than the connected lakes, suggesting an increased influence of evaporation (Figure 5.13). Strong trends over time were also visible in the isotope dataset (Figure 5.14). For the majority of the monitoring period, connected lakes had $\delta^{18}O$ that varied between 5 and 7‰. Connected lakes showed seasonal variations in $\delta^{18}O$ from March 2005 to August

2006, with higher ^{18}O values during summer then decreases during the autumn and winter. Seasonality was not as strong during 2007 as in previous years. ^{18}O values in connected lakes increased slightly throughout 2007 to reach a maximum during December 2007. A strong trend towards increasingly depleted ^{18}O conditions occurred between December and February 2008. This depleted signal was maintained throughout 2008 (Matthew Jones, pers. comm.).

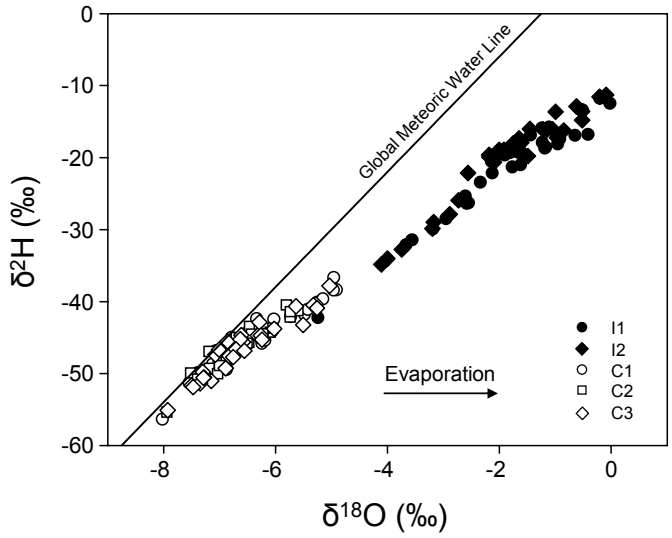

Figure 5.13 $\delta^{18}O$ and $\delta^{2}H$ values from connected lakes and isolated lakes I1 and I2, May 2005 – January 2008, shown against the Global Meteoric Water Line (GMWL), defined as $\delta^{2}H = 8\ \delta^{18}O + 10$ (Craig, 1961, in Darling et al., 2005).

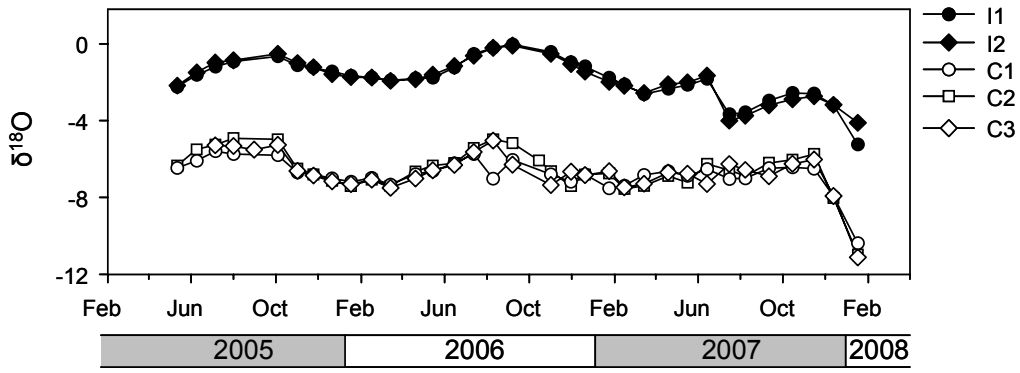

Figure 5.14 $\delta^{18}O$ values in connected lakes and isolated lakes I1 and I2, May 2005 - January 2008.

In isolated lakes I1 and I2, $\delta^{18}O$ values were higher than in connected lakes, reflecting increased concentration by evaporation compared to connected lakes. However, the seasonal cyclicity in $\delta^{18}O$ values was still visibile, tracking the $\delta^{18}O$ signal in the connected lakes. During 2005 and 2006 $\delta^{18}O$ values increased during spring and reached maxima of -0.03‰ during late summer 2006. $\delta^{18}O$ values decreased between August 2006 and March 2007. $\delta^{18}O$ values decreased abruptly between July and August 2007. Increases in $\delta^{18}O$ values were observed in both isolated lakes during the late summer and autumn 2007, which was followed by declines between November 2007 and January 2008. Isotope data were not available for Lake I3.

5.3.5 Isolated lake hydrology

In both isolated lakes I1 and I2, water depth was generally greater during 2007 than either 2005 or 2006 (Figure 5.15a and b). Changes in water depth were proportionally and absolutely greater in Lake I2 than in Lake I1 and increases in water depth tended to be slightly lagged in Lake I2 compared to Lake I1. In both lakes, water depth was lowest during summer 2005, but during 2006 the lowest water depth did not occur until October. From October 2006 until April 2007, water depth rose in both lakes. During 2007, water depth was greatest during spring, and then declined during the summer. Water depth during autumn 2007 was higher than in 2005 or 2006, and rose between November 2006 and February 2007 in both lakes. During winter 2005 and spring 2006, increases in water depth were greater in Lake I2 than in Lake I1. Standardised water depths in Lake I1 and Lake I2 were significantly positively correlated (Figure 5.16, $r_s = 0.680$, $p \leq 0.001$).

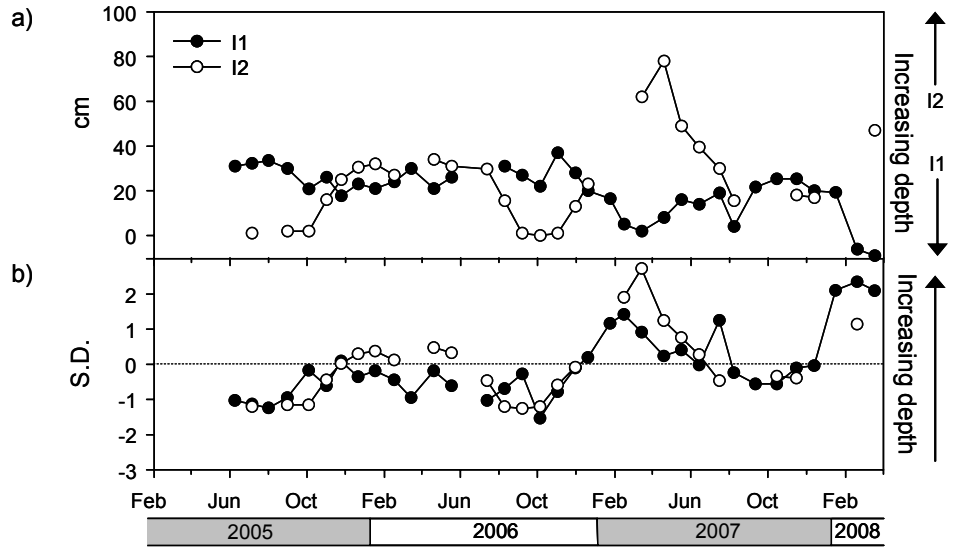

Figure 5.15 Water levels in isolated lakes I1 and I2, June 2005 - March 2008. a) For Lake I1, 'level' refers to measure of water surface below a fixed point, and for Lake I2, 'level' is the water depth at a fixed point. Note the direction of increases in water depth. b)Water depths expressed as the number of standard deviations (S.D.) from the mean, represented by the dashed line. The mean water level (± S.E.) and S.D. in square parentheses are: Lake I1: -19.32 (± 1.95) [11.56] cm; Lake I2: 24.60 (± 3.85) [19.62] cm.

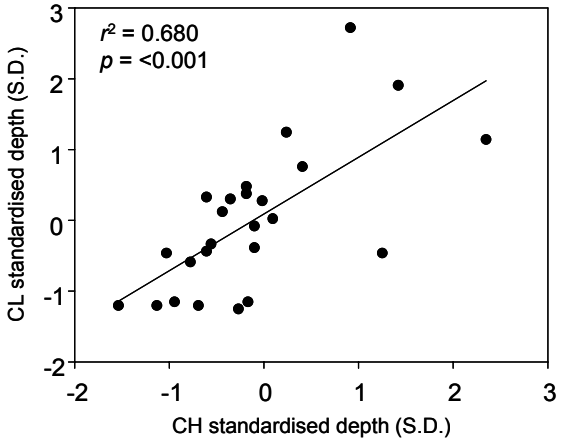

Figure 5.16 Correlation between standardised water depths in lakes I1 and I2, June 2005 - March 2008.

Historical water level data gathered in Lake I2 by S. Aitken (Figure 5.17) suggest a clear response to precipitation. Water level changes observed during the summer of 2007 were anomalous compared to other recorded years since 1995. Water depths have historically been greatest during the late winter and shallowest during the late summer and autumn. An increase of 107 cm in water depth occurred within only five days during June 2007, causing the highest recorded depth. Given the absence of substantial surface flows, groundwater and precipitation directly into the lake are likely to have accounted for this. Water depths for summer 2007 were at least 100 cm greater than comparable periods in other years, although the water depths during early 2008 were not unusual compared to previous years.

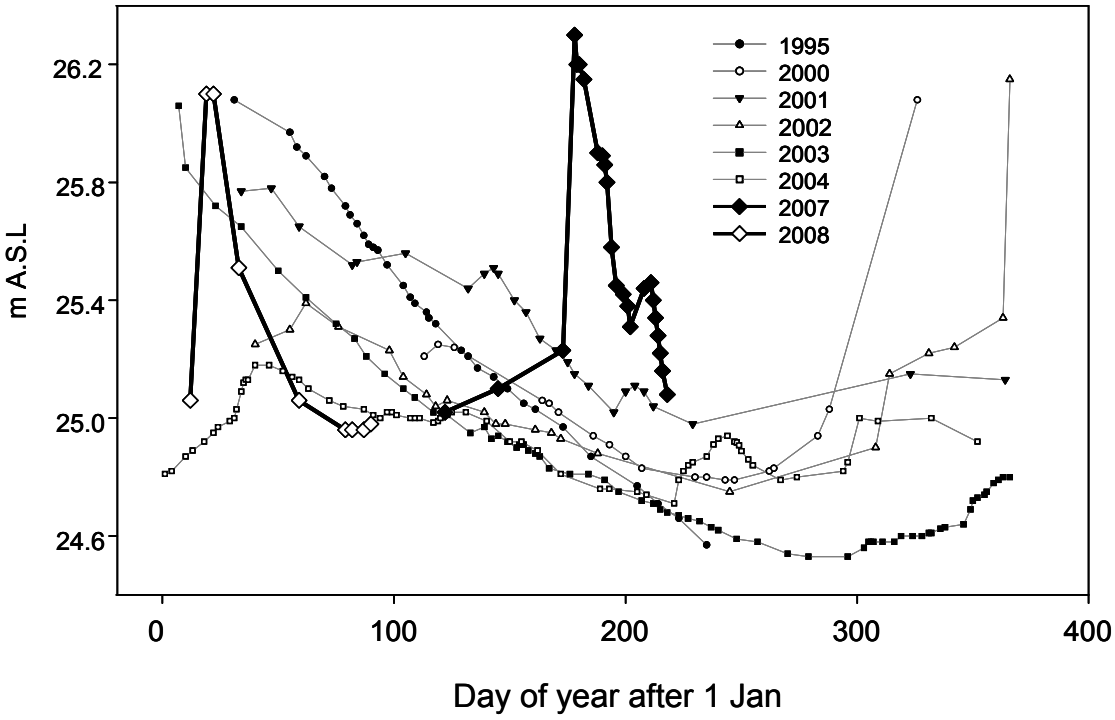

Figure 5.17 Water depths (m A.S.L.) observed in Lake I2 during previous years. Data for 2007 and 2008 are highlighted with bold lines. Measurements during 2007 were taken only during the wet period, and data for 2008 were available until April only. Data provided by Sandy Aitken (pers. comm.).

5.3.6 Hydrological summary

The location of the Attenborough Ponds at the confluence of the Rivers Trent and Erewash caused complex hydrological responses to the variations in rainfall during the monitoring period. These effects extended beyond connected lakes alone, with substantial changes in water level being observed in isolated lakes I1 and I2. The analysis of hydrology has identified a trend towards increasingly lotic conditions in connected lakes during 2007. Rainfall during 2005 and 2006 was comparable, whilst during 2007 precipitation was anomalously high during summer and early 2008. In the connected lakes, the lower $\delta^{18}O$ values during the summer of 2007 and sharp fall during November 2007 to January 2008 indicate increasingly wet conditions.

The hydrology of the connected lakes was strongly affected by the discharge of water to the River Trent. Three distinct regimes occurred during the monitoring period. Firstly, when only Ea is flowing, lakes C2 and C3 are effectively isolated and Lake C1 remains comparatively connected (Figure 5.11). These conditions were observed during much of 2005. The second regime occurred throughout 2006 and represents an intermediate level of connectivity. Flows were observed at MPO only during the winter and spring months, while during the summer, lakes C2 and C3 were effectively isolated. The third and most lotic regime occurs when the majority of flow exits the connected lakes via MPO. This state was observed largely during 2007, particularly during February to March 2007, July 2007 and January to March 2008. WRT during these periods was very short, frequently <3 days.

Despite being isolated from the River Erewash, changes in water depth in isolated lakes were consistent with those observed in connected lakes. Because of the absence of substantial surface inflows, groundwater and precipitation directly into the lakes likely to have been principally responsible for changes in water depth. Water depths were shallowest during the summer of 2005, greater during 2006 and anomalously high during 2007. $\delta^{18}O$ values supported the trend towards a wetter system during 2007. In connected lakes a sharp increase in water depth was observed after October 2006 in both lake I1 and I2. Water depths were rarely less than the long term average after January 2007. Changes in standardised and absolute water depth in Lake I2 were greater than in Lake I1, particularly during spring 2007.

Lake I2 water level changes were slightly lagged compared to those of Lake I1. Data for connected lakes suggests that groundwater and surface runoff are sufficient to induce substantial increases in water depth during 'wet' periods. However, during the highest water event in January 2008, overtopping of water was observed from C3 to I1 (Figure 5.18), which was not observed at any other time during the monitoring period.

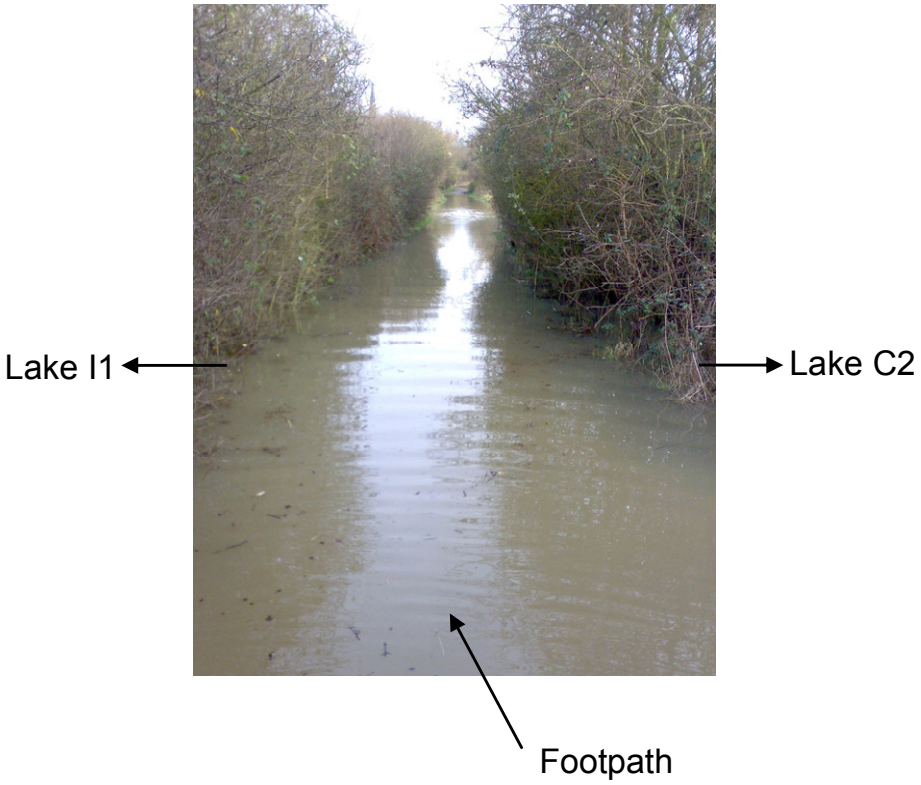

Figure 5.18 Water overtopping between lakes C2 and I1, January 2008. Lake C2 is on the right of the photograph, which was taken facing approximately north-east along the dividing embankment. The depth of the water on the embankment was approximately 30-40 cm.

5.4 Results - Inflow water chemistry

5.4.1 River Erewash and connected lakes outflow chemistry

Nutrients

Nutrient concentrations for Ebb (Figure 2.1, Figure 5.19, Table 5.1) were similar to Eb during the monitoring period. Significantly higher concentrations were only observed at Eb compared to Ebb between July 2006 and March 2008 for NH_4-N and NO_3-N concentration (Wilcoxon signed rank, z = -3.80 and -3.07 respectively, $p \leq 0.002$) and SiO_3 (z = -2.98, p = 0.002). The concentrations of inflowing nutrients measured at Eb were generally higher than in each of the connected lakes (Figure 5.19). TP concentrations at Eb were significantly higher than Lake C2 concentrations during the monitoring period (Table 5.2). Figure 5.19a shows that in-lake TP concentrations were greater than inflow concentrations during summer 2005 and 2006, but not 2007. TP concentrations were negatively correlated with River Erewash mean monthly discharge (r_s = -0.472, p = 0.003). SRP concentrations were generally greater during summer and autumn throughout the monitoring period at Eb than in the connected lakes (Figure 5.19b), although for the whole monitoring period were significantly higher at Eb. SRP concentrations were negatively correlated with Eb discharge (r_s = -0.572, p = <0.001). NH_4-N concentrations in inflowing water were highly variable (Figure 5.19c) and were significantly higher than in all connected lakes during the monitoring period (Table 5.2), particularly during the summers of 2005 and 2006. NH_4-N concentrations were not correlated with discharge at Eb. NO_3-N concentrations exceeded 15 mg L^{-1} during the winter of each year (Figure 5.19d) and were consistently and significantly higher than the in-lake concentrations (Table 5.2). NO_3-N concentrations at Eb and River Erewash discharge were negatively correlated (r_s = -0.477, p = 0.003). SiO_3 concentrations at Eb were positively correlated with River Erewash discharge (r_s = 0.608, p = 0.608) and were significantly greater than in-lake concentrations throughout the monitoring period (Table 5.2). SiO_3 concentrations were generally higher during the winter and lowest in the spring (Figure 5.19e).

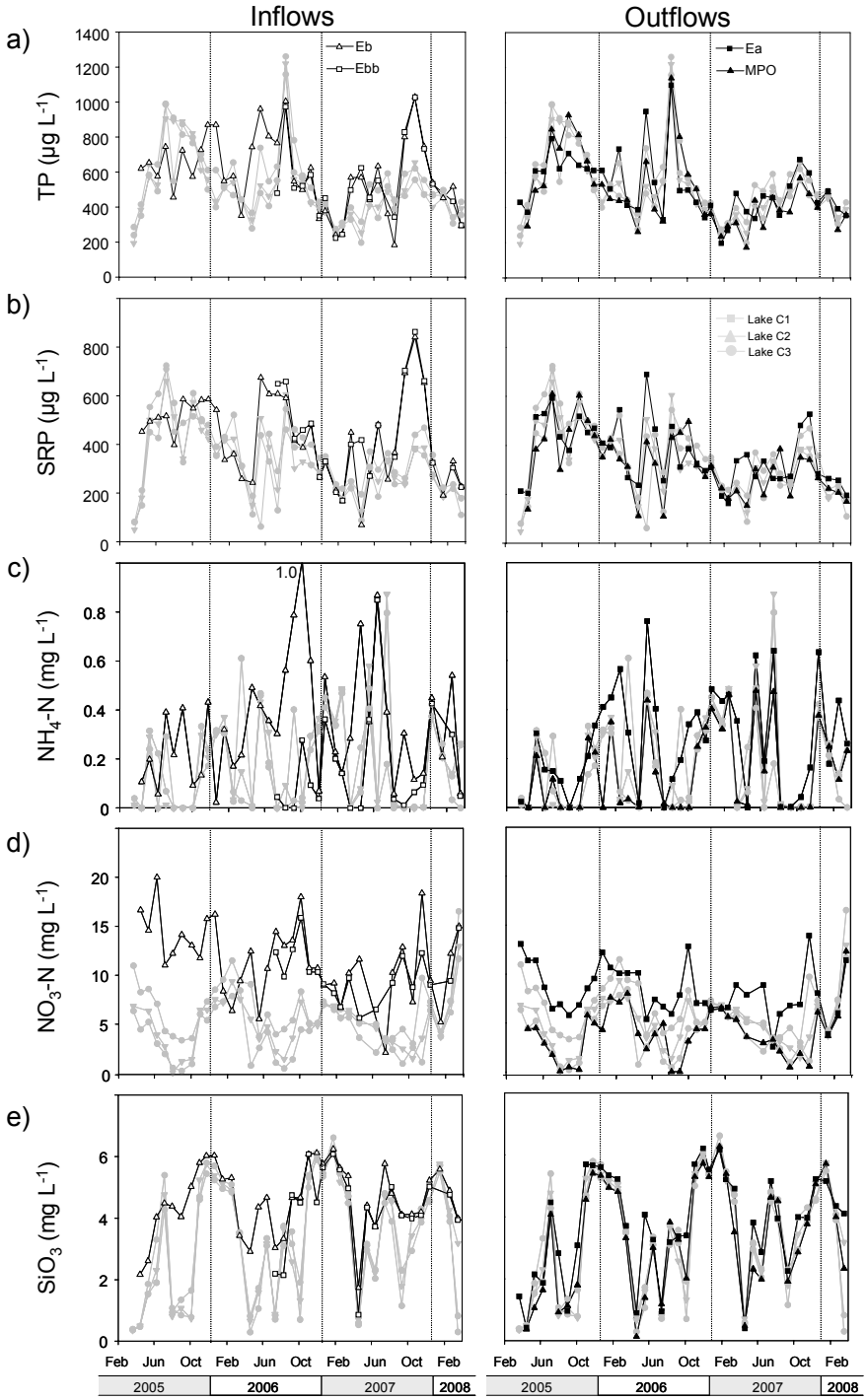

Figure 5.19 Chemical variables measured at inflows Eb (April 2005 - March 2008), and Ebb (July 2007 - March 2008) and outflows Ea and MPO (March 2005 - March 2008) to and from connected lakes: a) TP (µg L^{-1}); b) SRP (µg L^{-1}); c) NH_4-N (mg L^{-1}); d)NO_3-N (mg L^{-1}); e)SiO_3 (mg L^{-1}). In-lake concentrations are shown in grey symbols and lines.

Table 5.1 Summary statistics of nutrient concentrations observed at inflow site Ebb, July 2006 - March 2008.

Variable	Statistic			
	Mean	±1 S.E.	Min	Max
TP (µg L^{-1})	533	49	223	1027
SRP (µg L^{-1})	432	42	170	863
NH_4-N (mg L^{-1})	0.17	0.05	0	0.85
NO_3-N (mg L^{-1})	10.1	0.6	5.6	15.9
SiO_3 (mg L^{-1})	4.3	0.3	0.9	6.1

Some significant differences between in-lake nutrient concentrations and outflow concentrations were found (Figure 5.19, Table 5.2). Outflow concentration of TP and SRP were both lower than the in-lake concentrations, particularly at MPO. NH_4-N concentrations were significantly higher at both outflows than in-lake concentrations, although NO_3-N concentrations were significantly higher at Ea, but lower at MPO. In-lake SiO_3 concentrations were higher in the connected lakes than at the Ea outflow.

Table 5.2 Chemistry variables measured at inflow Eb and outflows Ea and MPO during the monitoring period (April 2005 - March 2008). *z*-scores for the Wilcoxon signed-rank test are given for the difference between each connected lake and each outflow for the duration of the monitoring period. Shaded *z*-scores indicate higher concentrations in the lake. Only significant values are given (*, $p \leq 0.05$, **, $p \leq 0.001$).

Variable	Statistic				Wilcoxon Signed Rank Test (*z*)		
	Mean	±1 S.E.	Min	Max	C1	C2	C3
Eb							
TP (µg L^{-1})	593	35	183	1030	-	-2.07**	-
SRP (µg L^{-1})	429	28	68	841	-2.00*	-2.94**	-2.80**
NH_4-N (mg L^{-1})	0.29	0.05	0.02	0.79	-3.14**	-2.53*	-2.34*
NO_3-N (mg L^{-1})	11.6	0.7	2.1	20.0	-4.75**	-4.93**	-5.08**
SiO_3 (mg L^{-1})	4.6	0.2	1.7	6.3	-4.79**	-4.94**	-5.11**
Ea							
TP (µg L^{-1})	522	29	196	1100	-2.37**	-	-
SRP (µg L^{-1})	367	20	163	689	-	-2.28*	-
NH_4-N (mg L^{-1})	0.34	0.04	0.02	1.00	-3.55**	-3.63**	-3.03**
NO_3-N (mg L^{-1})	11.6	0.7	2.1	20.0	-4.20**	-4.84**	-5.00**
SiO_3 (mg L^{-1})	3.8	0.3	0.4	6.2	-2.67*	-2.88**	-3.44**
MPO							
TP (µg L^{-1})	490	34	172	1139	-3.89**	-2.15*	-3.89**
SRP (µg L^{-1})	320	22	11	610	-3.10*	-2.20*	-
NH_4-N (mg L^{-1})	0.15	0.03	0.00	0.48	-	-2.58*	-3.06**
NO_3-N (mg L^{-1})	8.2	0.4	2.6	14.0	5.11**	-4.26**	-2.73**

Major ions and alkalinity

Major ion concentrations were similar between inflow sites Ebb and Eb (Table 5.3, Figure 5.20). However, Wilcoxon's signed rank test revealed a marginally significantly higher K^+ concentration at Eb compared to Ebb (z = -2.05, p = 0.04). Major ion concentrations at Eb were generally higher than those observed in the connected lakes (Table 5.4). The concentrations of major ions at Ea tended to be higher than in-lake concentrations, although outflow concentrations at MPO were generally higher than the in-lake concentrations in lakes C1 and C2.

Table 5.3 Statistics of major ion concentrations and total alkalinity (Tot. alk.) observed at inflow site Ebb, July 2006 - March 2008.

Variable	Statistic			
	Mean	±1 S.E.	Min	Max
Mg^{2+} (meq L^{-1})	1.47	0.12	0.55	2.31
Ca^{2+} (meq L^{-1})	1.96	0.15	0.91	3.15
Na^{+} (meq L^{-1})	3.37	0.29	1.22	5.88
K^{+} (meq L^{-1})	0.22	0.02	0.09	0.36
Cl^{-} (meq L^{-1})	2.69	0.21	0.80	4.18
Tot. alk. (meq L^{-1})	3.74	0.15	2.60	4.90

Table 5.4 Major ions and alkalinity measured at inflow Eb and outflows Ea and MPO during the monitoring period (April 2005 - March 2008). *z*-scores for the Wilcoxon signed-rank test are given for the difference between each connected lake and each outflow for the duration of the monitoring period. Shaded *z*-scores indicate higher concentrations in the lake. Only significant values are given (*, $p \leq 0.05$, **, $p \leq 0.001$). Tot. alk, total alkalinity.

Variable	Statistic				Wilcoxon Signed Rank Test (*z*)		
	Mean	±1 S.E.	Min	Max	C1	C2	C3
Eb							
Mg^{2+} (meq L^{-1})	1.45	0.07	0.23	2.62	-	-	-2.72**
Ca^{2+} (meq L^{-1})	2.23	0.13	0.94	4.36	-2.60*	-2.34*	-3.29**
Na^{+} (meq L^{-1})	3.34	0.18	0.63	4.91	-	-2.71**	-3.11**
K^{+} (meq L^{-1})	0.26	0.01	0.10	0.34	-2.01*	-2.34*	-3.15**
Cl^{-} (meq L^{-1})	3.01	0.15	0.78	5.44	-	-	-
Tot. alk. (meq L^{-1})	3.56	0.09	2.40	4.65	-	-2.36*	-2.43*
Ea							
Mg^{2+} (meq L^{-1})	1.45	0.07	0.41	2.42	-	-	-2.52*
Ca^{2+} (meq L^{-1})	2.15	0.11	0.91	3.38	-	-	-3.07**
Na^{+} (meq L^{-1})	3.34	0.18	0.63	4.91	-	-	-1.97*
K^{+} (meq L^{-1})	0.24	0.01	0.06	0.42	-	-	-2.51**
Cl^{-} (meq L^{-1})	2.79	0.20	0.04	6.62	-	-	-
Tot. alk. (meq L^{-1})	3.48	0.09	2.10	5.00	-	-	-
MPO							
Mg^{2+} (meq L^{-1})	1.36	0.06	0.67	2.23	-3.64**	-3.79**	-2.11*
Ca^{2+} (meq L^{-1})	2.02	0.10	3.35	0.70	-2.64**	-3.39**	-
Na^{+} (meq L^{-1})	3.00	0.18	1.03	5.24	-3.55**	-2.77**	-
K^{+} (meq L^{-1})	0.22	0.01	0.12	0.37	-3.05**	-2.29*	-
Cl^{-} (meq L^{-1})	2.82	0.20	0.95	5.78	-	-	-
Tot. alk. (meq L^{-1})	3.30	0.10	1.90	4.30	-2.03*	-	-

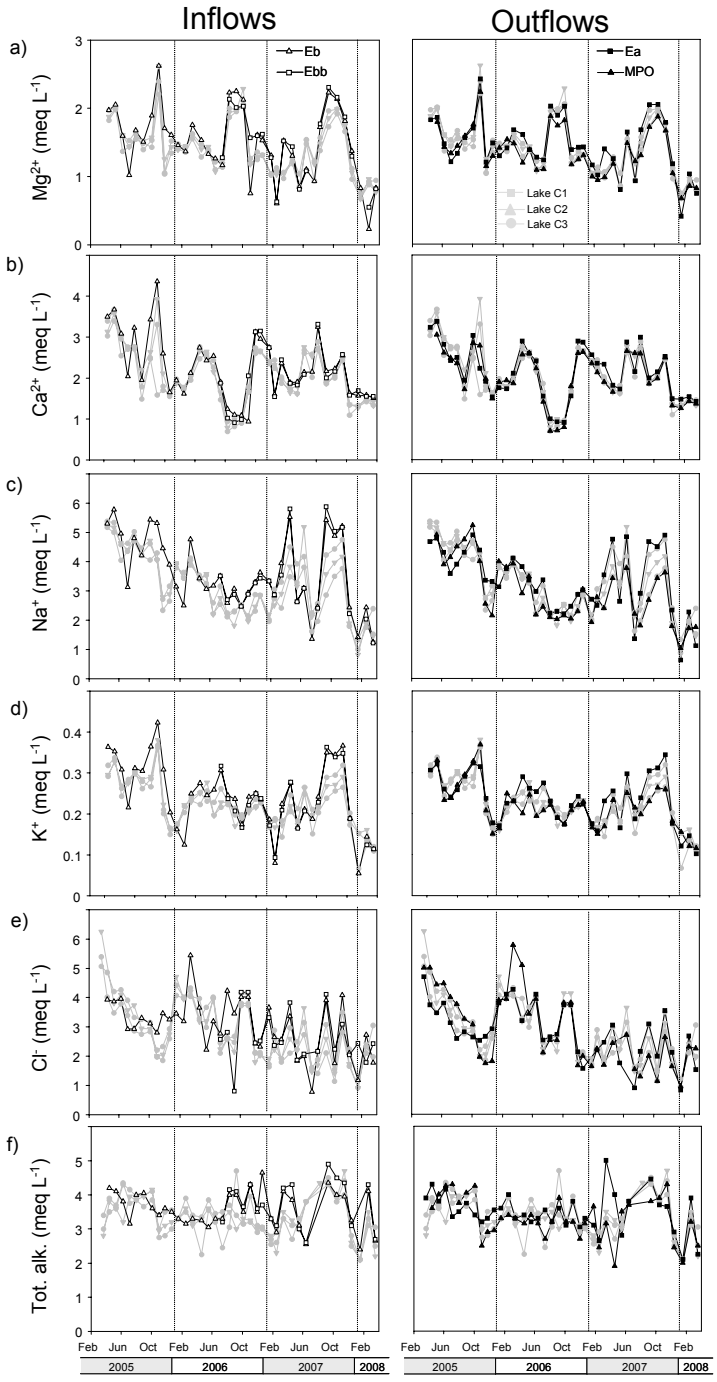

Figure 5.20 Major ions and total alkalinity measured at inflows Eb (April 2005 - March 2008), and Ebb (July 2007 - March 2008) and outflows Ea and MPO (March 2005 - March 2008) to and from connected lakes: a, Mg^{2+} (meq L^{-1}); b, Ca^{2+} (meq L^{-1}); c, Na^{+} (meq L^{-1}), d, K^{+} (meq L^{-1}); e, Cl^{-} (meq L^{-1}) and f, total alkalinity (meq L^{-1}). In-lake concentrations are shown in grey symbols and lines.

5.4.2 Lake I3 inflow (BPI) chemistry

All nutrient concentrations, except NH_4-N, were significantly higher in the inflowing water at site BPI (Figure 2.1, page 74) compared with in Lake I3 (Table 5.5). Little seasonality was observed in the concentrations of nutrients measured at BPI (Figure 5.21). No distinct interannual changes in nutrient concentrations were evident except for NH_4-N concentrations, which were lower throughout 2007 than in 2005 and 2006. Interannual differences and seasonal patterns in the concentrations of major ions and nutrients were not found, although temporal trends in Ca^{2+}, Na^{+}, and K^{+} were similar during the monitoring period. The concentrations of most major ions at BPI briefly increased during the winter of 2005. Smaller increases in in-lake concentrations were observed at the same time.

Table 5.5 Nutrients, major ions and alkalinity measured at inflow BPI during the monitoring period (June 2005 - March 2008). *z*-scores for the Wilcoxon signed-rank test are given for the difference between BPI and Lake I3 for the duration of the monitoring period. All differences indicate that concentrations were higher at inflow BPI than in Lake I3. Only significant values are given (, $p \leq 0.001$). Tot. alk, total alkalinity.**

Variable	Statistic				Wilcoxon Signed rank test (*z*)
	Mean	±1 S.E.	Min	Max	
TP (µg L^{-1})	552	36	200	916	-4.90**
SRP (µg L^{-1})	475	31	210	790	-5.09**
NH_4-N (mg L^{-1})	0.09	0.03	0.00	0.70	-
NO_3-N (mg L^{-1})	5.3	0.3	0.1	7.3	-4.94**
SiO_3 (mg L^{-1})	4.1	0.1	2.5	6.5	-5.09**
Mg^{2+} (meq L^{-1})	1.69	0.10	0.62	3.49	-3.78**
Ca^{2+} (meq L^{-1})	2.30	0.13	0.34	4.03	-4.10**
Na^{+} (meq L^{-1})	1.55	0.12	0.15	4.54	-
K^{+} (meq L^{-1})	0.88	0.01	0.03	0.18	-
Cl^{-} (meq L^{-1})	1.86	0.22	0.16	6.76	-
Tot. alk. (meq L^{-1})	4.28	0.16	1.00	5.35	-4.32**

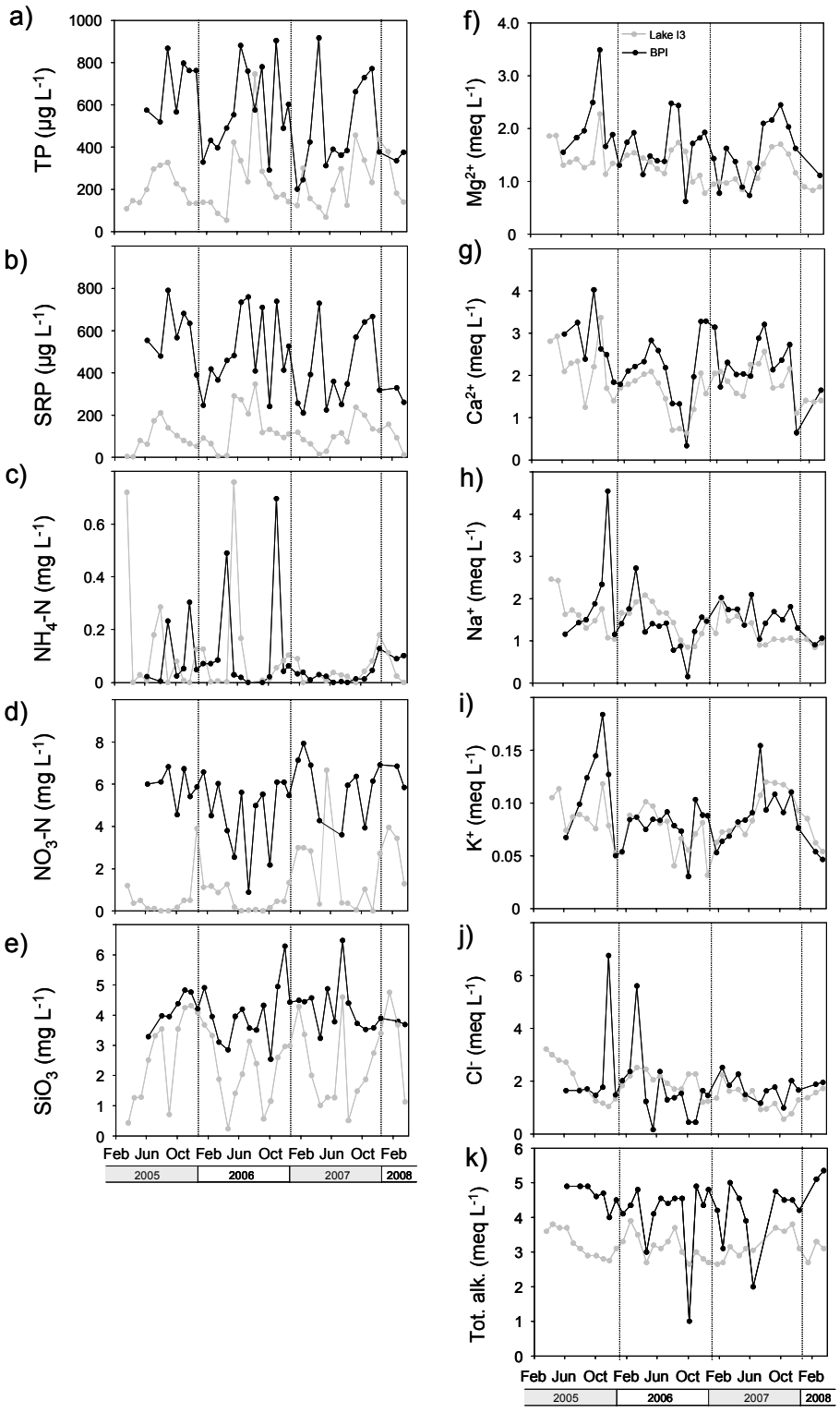

Figure 5.21 Chemical variables measured at BPI (June 2005 - March 2008) and in Lake I3 (March 2005–March 2008): a) TP (µg L^{-1}); b) SRP (µg L^{-1}); c) NH_4-N (mg L^{-1}); d) NO_3-N (mg L^{-1}); e) SiO_3 (mg L^{-1}); f) Mg^{2+} (meq L^{-1}); g) Ca^{2+} (meq L^{-1}); h) Na^{+} (meq L^{-1}); i) K^{+} (meq L^{-1}); j) Cl^{-} (meq L^{-1}); k) total alkalinity (meq L^{-1}). In-lake samples from Lake I3 are shown in grey.

5.5 Results - Nutrient budget for connected lakes

5.5.1 Mass balance

An estimated 22.8 t of TP entered the connected lakes between March 2005 and February 2006. For equivalent periods during 2006 and 2007, 29.5 and 28.2 t of TP entered the lakes. Estimated SRP loading in the same periods was 20.6, 24.0 and 24.3 t. More TP was retained in the connected lakes during 2006 (6.1 t) than 2005 (4.8 t loss, Figure 5.22a). Similarly, SRP was lost during 2005 (3.4 t) and retained during 2006 (3.4 t). No clear seasonality in retention and loss of TP or SRP was observed. High estimates of discharge from the lakes during 2007 show that more TP and SRP was lost than entered the lakes.

NH_4-N loading varied between years. 9.7 t flowed into the connected lakes during 2005, although in 2006 and 2007 this rose to 27.2 and 24.5 t respectively. NH_4-N was lost from the connected lakes during the 2005 (4.4 t) and 1.7 t was retained during 2006 (Figure 5.22). High estimates of loss during 2007 were associated with high discharge from the connected lakes. Large quantities of NH_4-N were also released from the connected lakes during January and March 2008.

NO_3-N inputs during 2005, 2006 and 2007 were 549, 701 and 760 t (1st March to 28th February). Inputs of NO_3-N to the connected lakes were substantially higher than the outflows during 2005 and 2006, suggesting that large quantities of NO_3-N were retained within the lakes (Figure 5.22). NO_3-N was only released from the lakes in substantial quantities during April 2005, February and March 2006 and February 2007. During 2007, large quantities of NO_3-N was lost, and the lakes acted as a NO_3-N sink during August, September and December 2007. Large losses of NO_3-N occurred during January and March 2008 although during February 2008 the connected lakes appeared to act as a sink of NO_3-N.

SiO_3 inputs varied among the years of the monitoring period. Between April 2005 and March 2006 213 t flowed into the connected lakes. During 2006 and 2007, estimated inputs of SiO_3 were 342 and 320 t. The retention and release of SiO_3 varied throughout the monitoring period (Figure 5.22e). Greater quantities of SiO_3 appear to

have been retained within the connected lakes during 2006 than 2005. For most of 2006, connected lakes retained SiO_3. Retention of SiO_3 was strongest during April 2006, and then decreased throughout the summer. Retention of SiO_3 occurred during the winters of 2005 and 2006. Large quantities of SiO_3 were estimated to have been lost from the connected lakes during 2007. SiO_3 losses were large between January and March 2008, reflecting the high discharge at the outflow site.

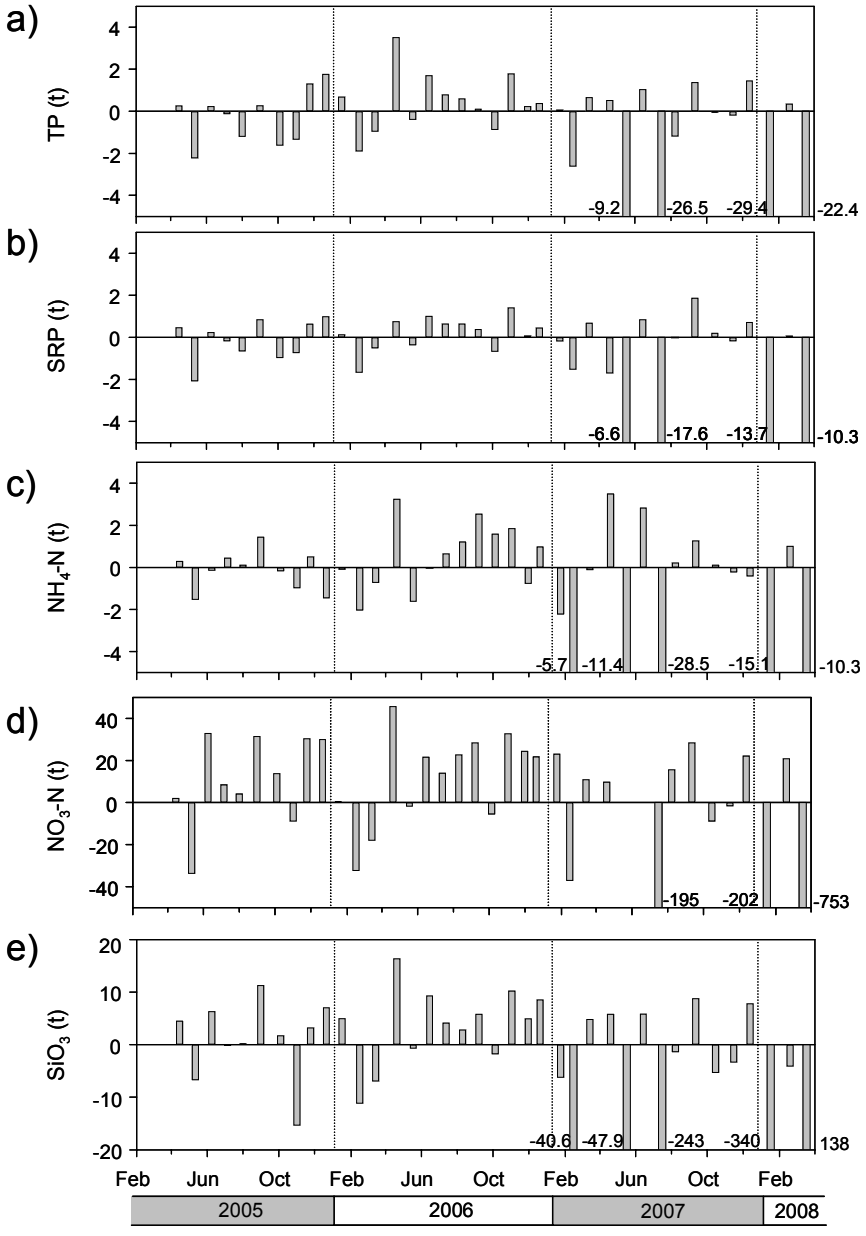

Figure 5.22 Retention (defined as input - loss) of nutrients within connected lakes, April 2005 to March 2008: a, TP; b, SRP; c, NH_4-N; d, NO_3-N; e, SiO_3, all calculated as tonnes (t) on a monthly basis. Negative values indicate loss of nutrient.

5.5.2 Vollenweider modelling

The Vollenweider model used to predict in-lake TP concentrations showed reasonable agreement with the observed data (r_s = 0.50, p = 0.001, Figure 5.23) suggesting that temporal changes were accurately modelled. However, predicted mean annual TP concentrations were less than the observed values for each year of the monitoring period (Figure 5.24). The model underestimated summer in-lake TP concentrations during 2005 and 2006. Observed concentrations were over 350 µg L^{-1} higher than the predicted concentration between July and October. Similar discrepancies were not so evident during 2007. During the winter and spring of all years, the model was more accurate and predicted TP concentrations were frequently within 100 µg L^{-1} of the measured value. On average, estimates were 37% below the observed TP concentration. The predicted mean annual TP concentrations were then used to model chlorophyll-*a* concentrations. Mean observed annual chlorophyll-*a* concentrations were always lower than the modelled value (Figure 5.24b). The difference between modelled and observed mean annual chlorophyll-*a* concentrations was greatest during 2007 (32 µg L^{-1}).

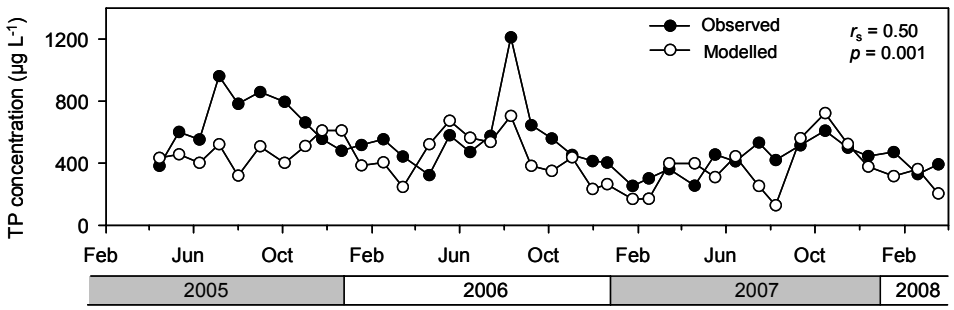

Figure 5.23 Observed and modelled in-lake TP concentrations for April 2005 to March 2008.

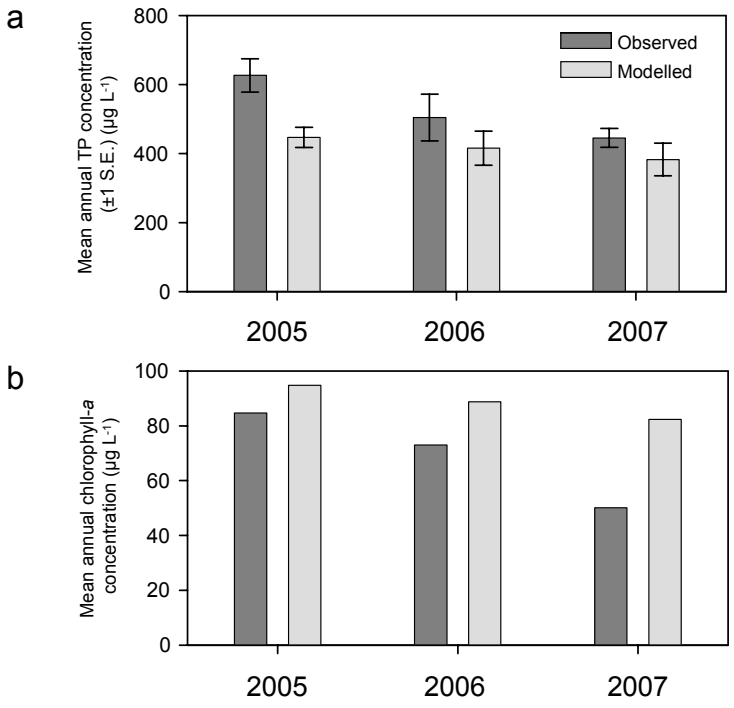

Figure 5.24 Comparison of mean annual in-lake TP and chlorophyll-a concentrations with modelled estimates. Chlorophyll-*a* estimates are based on the mean annual in-lake TP concentration.

Since the Vollenweider model predicted the measured TP concentrations with reasonable accuracy, parameters of discharge and nutrient loading were altered to investigate the possible response of the connected lakes to changes in discharge and reductions in nutrient loading. The annual mean TP concentrations predicted by the model were then used to estimate annual mean chlorophyll-*a* concentrations. These insights are not intended to be accurate predictions of future scenarios, but to investigate the sensitivity of in-lake TP and chlorophyll-*a* concentrations to hydrological and nutrient loading changes.

In order to do this, scenarios of reductions in inflow and reductions in nutrient loading were created. Discharge scenarios were based on reductions of River Erewash monthly inflow, measured at site Ea, to 50%, 75% and 95% of the discharge observed during the monitoring period. These were chosen to investigate how sensitive in-lake TP concentrations were to variations in discharge. These scenarios were run using the observed nutrient concentrations measured at Ea during the monitoring period. The

same model was then run with altered concentrations of TP in the inflowing water in order to explore how nutrient loading may affect the in-lake TP concentrations. The scenarios of nutrient loading assumed that the discharge of the River Erewash would not change from the values observed during the monitoring period. Nutrient reduction scenarios (10, 25 and 50% reduction from concentrations observed during the monitoring period) were based on published examples of decreases in P concentrations for lake restoration (Osborne, 1981; Köhler et al., 2005; Phillips et al., 2005; Rip et al., 2005; Søndergaard et al., 2005). In comparison to the published examples, the scenarios chosen here represent a conservative estimate of possible nutrient reduction. A final scenario was created using the predicted changes in inflow occurring from the diversion of the River Erewash. Summer flows (1 May to 30 September) and winter flows (1 October to 31 March) are expected to be reduced to 6% and 8% of pre-diversion discharges (HR Wallingford, 2006). This scenario adjusted summer and winter discharges to the predicted values and assumed that there would be no change in inflowing nutrient concentrations compared to those observed at site Ea during the monitoring period.

The results of the different scenarios suggest that the in-lake TP and chlorophyll-*a* concentrations are not responsive to discharge (Figure 5.25 and Figure 5.26). At the greatest reduction in discharge into the connected lakes (to 5% of observed values) mean annual in-lake TP concentration was only reduced by less than 1 µg L^{-1} when the model was run with observed TP concentrations. Reductions in chlorophyll-*a* concentration were negligible. A much greater response in in-lake TP and chlorophyll-*a* to reductions in nutrient loading was observed. Reducing the inflowing nutrient concentrations by 10% each month reduced the in-lake TP concentration by approximately 45 µg L^{-1}, and mean chlorophyll-*a* by 9 µg L^{-1} during each of the years modelled.

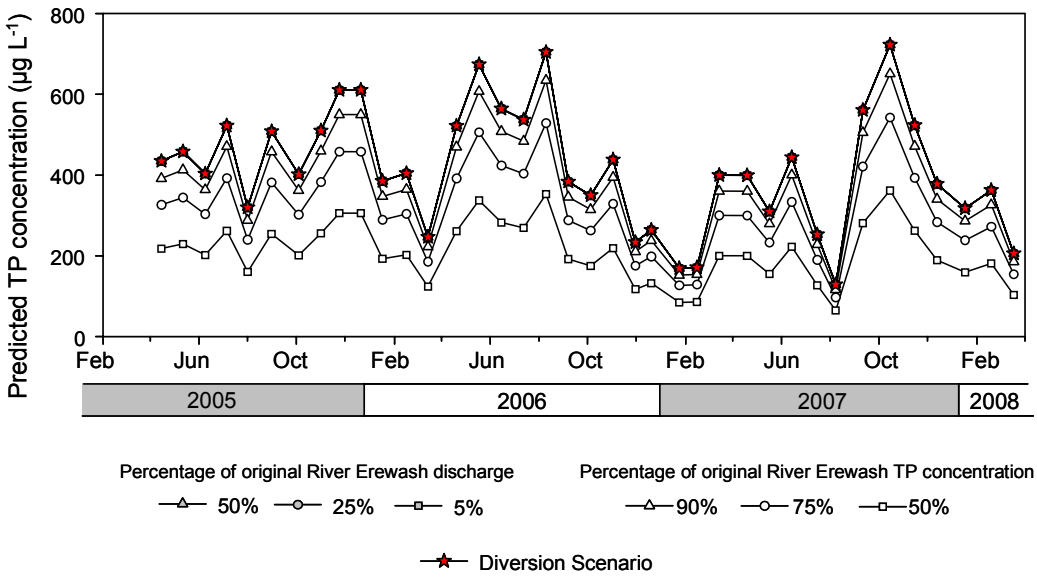

Figure 5.25 Predicted mean in-lake TP concentrations for connected lakes, April 2005 to March 2008, at different scenarios of nutrient loading and River Erewash discharge.

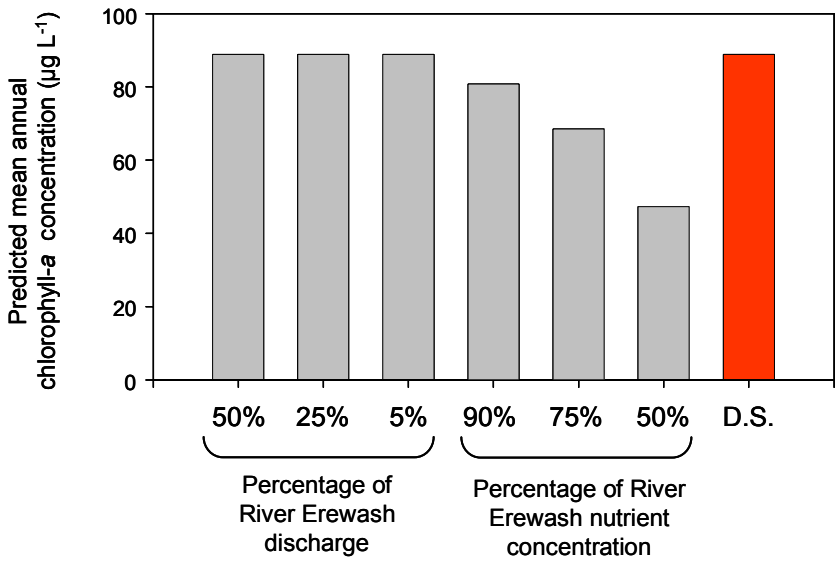

Figure 5.26 Predicted mean annual chlorophyll-*a* concentrations in connected lakes, based on scenarios of alterations in River Erewash flow and reduced nutrient loading. Chlorophyll-*a* concentration was predicted as a function of mean annual TP concentration using the equation shown in Figure 5.24. D.S. = diversion scenario.

5.6 Results - Interannual variability in connected and isolated lakes

5.6.1 Total phosphorus

TP concentrations (Figure 5.27, originally discussed in Chapter 3) in all connected lakes were generally lower between March and October 2007 than the same period in 2005 and 2006. From October to February the TP concentrations were slightly lower during 2006 than other years. Although TP concentrations were generally lower in 2007 than the other years, the same temporal sequence was observed in all years. The summertime increase in TP concentrations was not strong in the connected lakes during 2007, particularly in Lake C1 where TP declined by over 200 µg L^{-1} between July and August 2007. In lakes C2 and C3, the highest TP concentrations occurred later in the year during 2007 than in 2005 or 2006. In Lake I1, TP concentrations were similar each year until early June. Summertime maximum TP was substantially lower during 2007 than other years. In Lake I2, the maximum TP concentration during 2007 also was less than 2005 and 2006. Concentrations during the rest of the year were similar each year. In Lake I3, little similarity was observed between each of the three years of the monitoring period. The summertime maximum TP concentration was greater than during 2005 than 2006 and 2007. Autumn and winter TP concentrations in Lake I3 were higher than during 2005 and 2006.

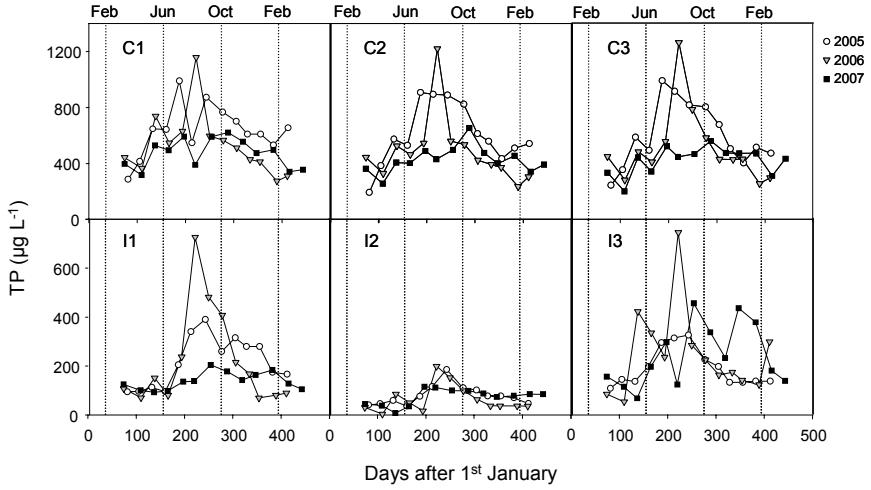

Figure 5.27 TP concentrations (µg L^{-1}) plotted against the number of days after 1st January for each year and each lake. 'Year' refers to March to February (31st March for 2007). Note the difference in *y*-axis scales.

5.6.2 Soluble reactive phosphorus

In connected lakes, SRP concentrations (Figure 5.28) generally increased during the spring and early summer and declined from late summer to winter. SRP data were originally discussed in Chapter 3. In Lake C1 SRP concentrations during 2007 were similar to those observed during 2006 and less than 2005 concentrations. The maximum SRP concentration occurred later in the year during 2007. Variations in SRP concentrations in lakes C2 and C3 were similar throughout the monitoring period. In both lakes, SRP concentrations were lower during 2007 than either 2005 or 2006, particularly during spring and late summer.

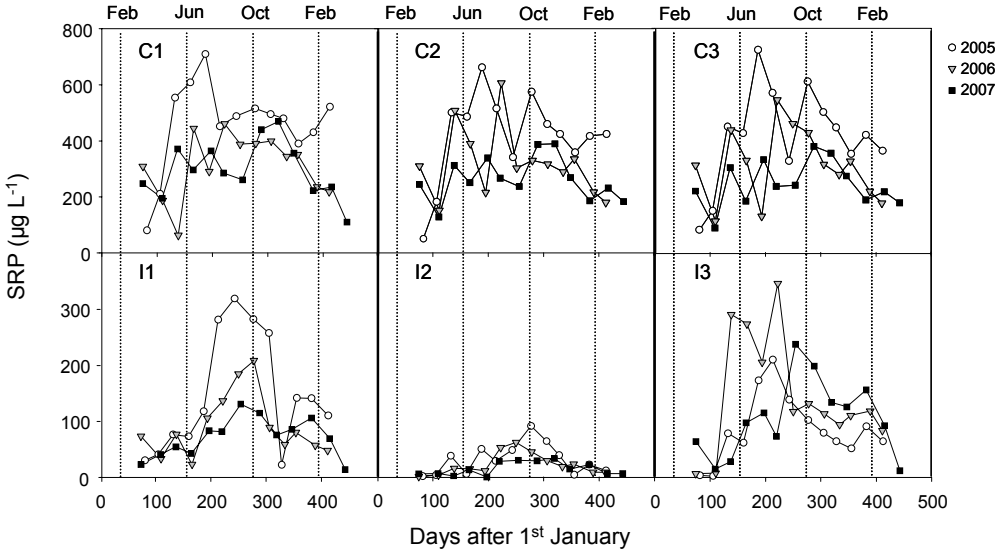

Figure 5.28 SRP concentrations (µg L^{-1}) plotted against the number of days after 1st January for each year and each lake. 'Year' refers to March to February (31st March for 2007). Note the difference in *y*-axis scales.

In Lake I1, SRP concentrations were generally lower during 2007 than 2005 or 2006, although the same seasonality was observed. In Lake I2, summer SRP concentrations during 2007 were also less than during 2005 or 2006 during summer. In Lake I3 SRP concentrations varied between years. SRP concentrations were highest until August in 2006. From August to March, SRP concentrations were highest during 2007. Similarly to the other isolated lakes, SRP concentrations increased from March until September and then declined until February. Maximum concentrations of SRP in Lake I3 occurred later in 2007 than 2005 or 2006.

5.6.3 Nitrate

NO_3-N data were originally presented in Chapter 3. In connected lakes, NO_3-N concentrations (Figure 5.29) did not clearly differ between years. In all years, NO_3-N concentrations declined during spring and early summer after which concentrations increased throughout the remainder of each year. Rapid rises in NO_3-N concentration were observed in all connected lakes between January and March 2008. In the isolated lakes, NO_3-N concentrations were greatest during winter and lowest during the summer. NO_3-N concentrations during 2007 were similar to those observed during 2005 and 2006, although rapid rises were observed during autumn and winter 2007. Overtopping from Lake C2 probably supplied some NO_3-N to Lake I2 during January 2008 (Figure 5.18, page 245).

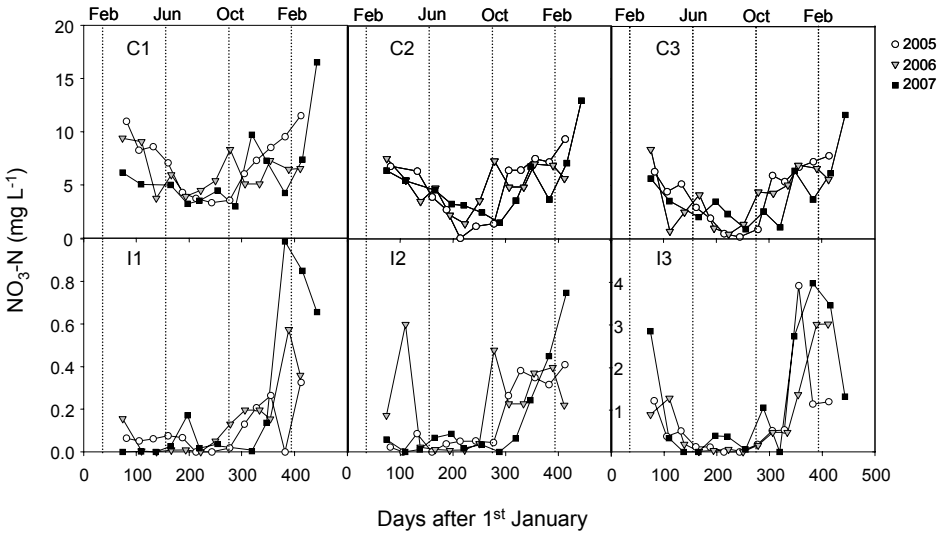

Figure 5.29 NO_3-N concentrations (mg L^{-1}) plotted as numbers of days after 1st January for each year and each lake. 'Year' refers to March to February (31st March for 2007). Note the difference in *y*-axis scales.

5.6.4 Ammonia

In connected lakes, NH_4-N concentrations (Figure 5.30, originally presented in Chapter 3) observed during 2007 did not differ substantially from those of 2005 or 2006. A similar seasonality was observed between lakes each year, with concentrations being highest during the spring and winter months. NH_4-N concentrations were higher in lakes C2 and C3 than C1. Little difference in NH_4-N concentrations between years was observed in Lake C1. In lakes C2 and C3, NH_4-N

concentrations high NH_4-N concentrations were observed in July 2007 although this was not observed in either 2005 or 2006. In all connected lakes, increases in NH_4-N concentration during winter occurred approximately one month later than in 2005 and 2006 although the increase was of a similar magnitude each year (approximately 0.5 mg L^{-1}).

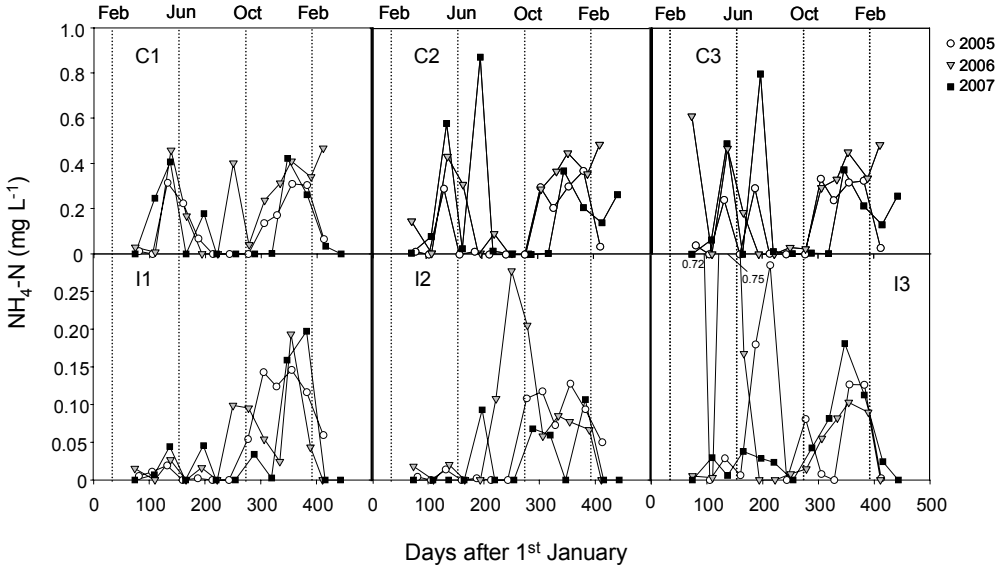

Figure 5.30 NH_4-N concentrations (µg L^{-1}) plotted against the number of days after 1st January for each year and each lake. 'Year' refers to March to February (31st March for 2007). Note the difference in *y*-axis scales.

Interannual patterns were more consistent in Lake I1 than in other isolated lakes. Concentrations of NH_4-N were highest during November to January each year. NH_4-N concentrations were slightly higher during spring and early summer during 2007 and lower in autumn than in 2005 and 2006. In Lake I2, NH_4-N concentrations in 2005 and 2007 were similar although were substantially higher during summer 2006. In Lake I3, interannual differences were observed in NH_4-N concentrations during the spring and summer periods. During 2005 and 2006, sharp increases were observed in March and May respectively, although this did not occur in 2007 when concentrations remained comparatively low during the same period. From October to February, changes in NH_4-N concentration were more consistent between each lake.

5.6.5 Silica

Amongst connected lakes seasonality in SiO_3 concentrations was generally consistent between years (Figure 5.31, originally discussed in Chapter 3). This was defined by

reductions in SiO_3 concentrations during spring and late summer although there were differences between the timing and degree of decrease observed each year particularly during the summer. SiO_3 concentrations in all connected lakes generally decreased in spring and then increased throughout summer. Whilst during July and August SiO_3 concentrations decreased in 2006, during 2005 and 2007 SiO_3 concentrations increased during the same period. The summer decline in SiO_3 concentration was not as large or prolonged during 2007 as in 2005 or 2006. During 2007, increases in SiO_3 concentrations were not as rapid as in 2005 or 2006 although the maximum concentration was similar in all three years (approximately 5.5 mg L^{-1}). Lakes I1 and I2 did not show such strong seasonality. SiO_3 concentrations in lakes I1 and I2 were generally higher during 2006 than 2005 or 2007. In Lake I1 SiO_3 concentrations remained relatively constant throughout 2005, although during 2006 a large increase in SiO_3 concentration occurred from July to November. SiO_3 concentrations during 2007 were generally similar to those observed during 2005 after decreasing from the large increase observed during 2006. A similar pattern occurred in Lake I2 although the magnitude of the increase in SiO_3 concentration during summer 2006 was less than that observed in Lake I1. SiO_3 concentrations in Lake I3 showed similar seasonality to connected lakes. During both the spring and summer maxima, SiO_3 concentrations were slightly greater in 2007 than 2005 or 2006.

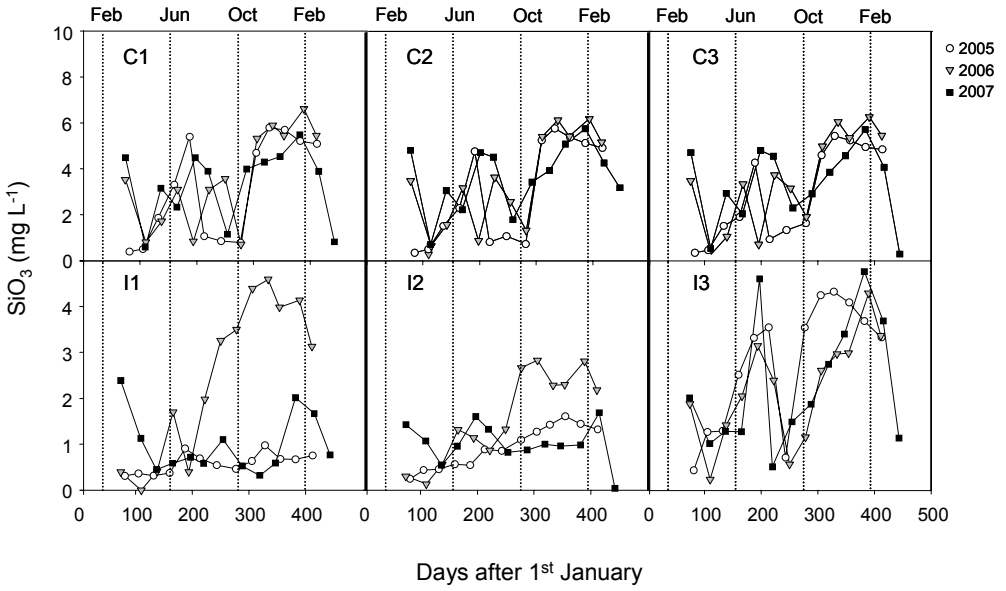

Figure 5.31 SiO_3 concentrations (mg L^{-1}) plotted as numbers of days after 1st January for each year and each lake. 'Year' refers to 1st March to 28th February (31st March for 2007). Note the difference in *y*-axis scales.

5.7 Results - Ecological response

5.7.1 Phytoplankton

Chlorophyll-*a* concentration

In the connected lakes, chlorophyll-*a* concentrations (originally discussed in Chapter 3, Figure 5.32) were strongly seasonal during 2005 and 2006. The magnitude and timing of changes in chlorophyll-*a* concentration were similar between each year. Substantial differences in chlorophyll-*a* concentrations were observed during 2007 compared to other years. Chlorophyll-*a* concentrations did not rise as rapidly during the summer in 2007 as in other years. In Lake C1, the chlorophyll-*a* concentration increased between May and June 2007, a month earlier than the increase observed in 2005 and 2006. In all connected lakes, chlorophyll-*a* concentrations decreased during mid-summer. During 2007, the highest chlorophyll-*a* concentration in all connected lakes occurred during September, a month later than during 2005 and 2006.

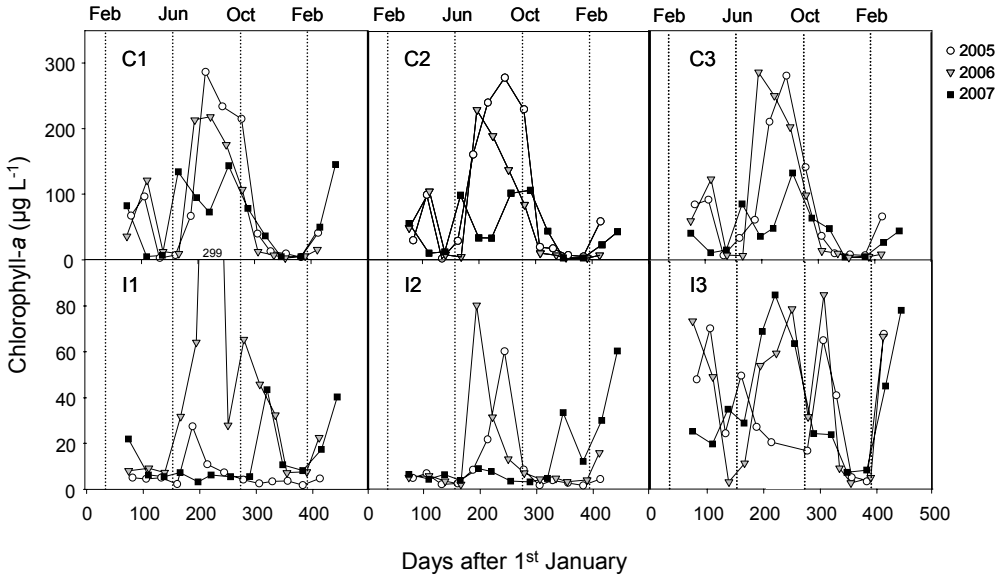

Figure 5.32 Chlorophyll-*a* concentrations (µg L^{-1}) plotted against the number of days after 1st January for each year and each lake. 'Year' refers to March to February (31st March for 2007). Note the difference in *y*-axis scales.

Chlorophyll-*a* concentrations in isolated lakes I1 and I2 were markedly different in 2007 from those observed in 2005 and 2006. In Lake I1, chlorophyll-*a* concentrations were low throughout the early summer during 2007. During 2005 and 2006 the highest chlorophyll-*a* concentration occurred during June and August respectively. In 2007, the highest recorded chlorophyll-*a* concentration occurred during November. The

highest chlorophyll-*a* concentration occurred during 2006 and was around seven times higher than that observed in 2007. In Lake I2, chlorophyll-*a* concentrations did not increase substantially during summer 2007 as in 2005 or 2006. The highest concentrations of chlorophyll-*a* occurred much later in the year during 2007 (November) compared to 2005 (July) and 2006 (August). Increases in chlorophyll-*a* concentration also occurred from January to March 2008. In Lake I3 chlorophyll-*a* concentrations were similarly variable during 2007 as during 2005 and 2006. During spring 2005 and 2006, chlorophyll-*a* concentrations fell, although increased during 2007. Maximum chlorophyll-*a* concentrations occurred at different times each year (March 2005, November 2006 and August 2007) and were similar among each year. Low concentrations of chlorophyll-*a* concentrations occurred during December and January each year.

Phytoplankton community composition

Phytoplankton community composition was compared between lakes in section 3.8. Axis 1 scores in connected lakes (Figure 5.33a, CA plot shown in Figure 3.49, page 177) were similar during each year of the monitoring period. Scores typically became higher in early summer then decreased between July and September. Low axis 1 scores were associated with chlorophytes, centric diatoms and the cyanobacterium *Microcystis* sp. Scores then increased during the autumn and winter, reflecting the increasing importance of cryptophytes and pennate diatoms in the winter phytoplankton community. In Lake C1, little difference was observed in axis 1 scores between each year throughout the monitoring period. In Lakes C2 and C3, axis 1 scores appeared to be slightly less homogenous between years although the same seasonal trend was observed in all years. Axis 1 scores during 2007 were slightly less than those during 2005 and 2006 during spring, suggesting a greater proportion of diatoms and cryptophytes in the phytoplankton community. For the remainder of the year, little consistent difference in axis 1 scores was observed. Brief increases in axis 1 scores were observed during the summer in both lakes C2 and C3, which may be a result of the small biovolumes of cyanobacteria observed during summer 2007.

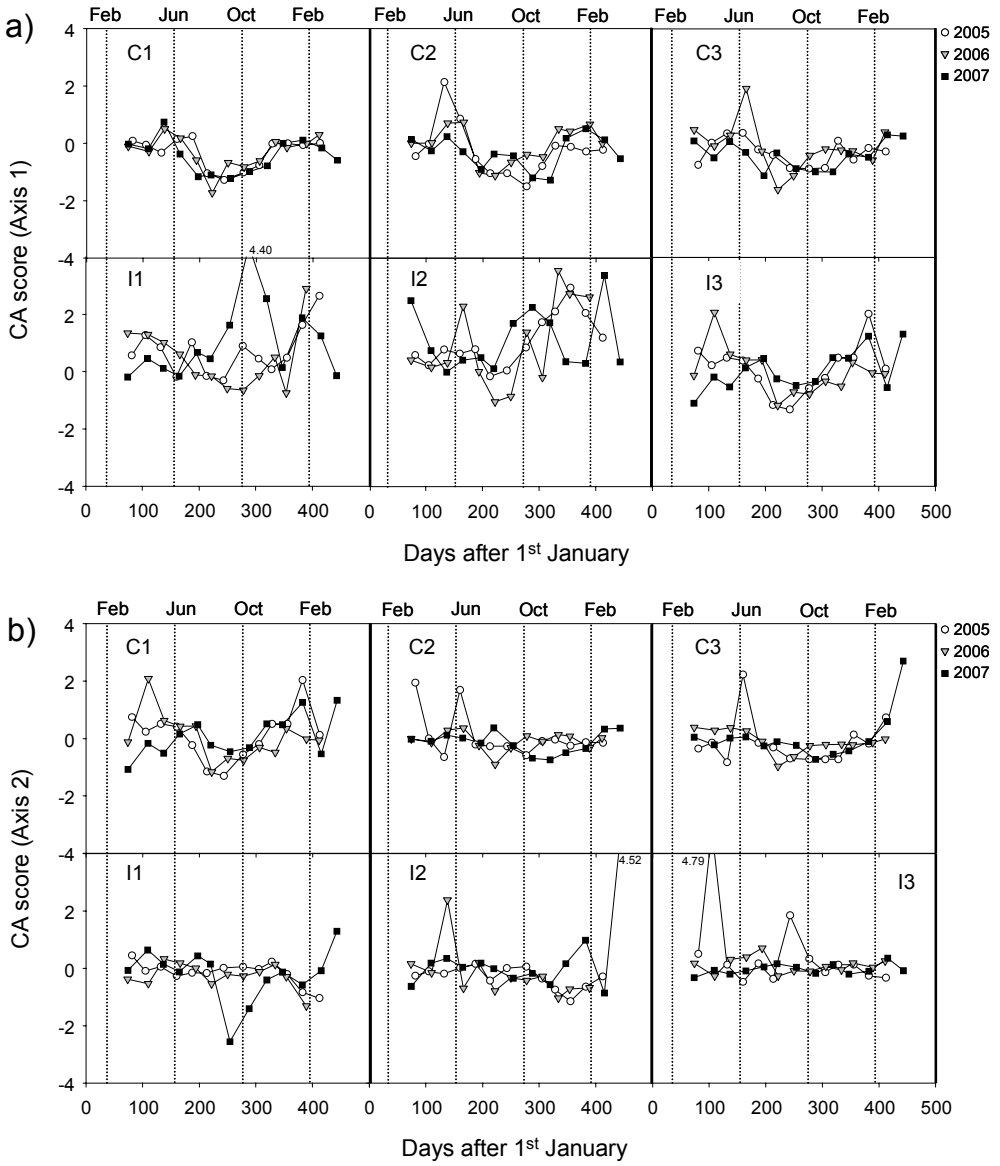

Figure 5.33 CA phytoplankton sample scores for a) axis 1, and b) axis 2. Scores plotted against the number of days after 1st January for each year and each lake. 'Year' refers to March to February (31st March for 2007). The species CA plot can be found in Figure 3.49 on page 177.

Phytoplankton axis 2 scores (Figure 5.33b) appeared to be more variable in Lake C1 than in either of Lake C2 or C3. During summer 2007, axis 2 scores were generally lower than during 2005 and 2006. In lakes C2 and C3, axis 2 scores were similar between each year, other than in early summer and spring 2005 when a relative abundance in small flagellates resulted in higher axis 2 scores. Increases in axis 2 scores in March 2008 in Lake C3 reflected the increase in *Mallomonas* sp.

Phytoplankton axis 1 scores differed amongst isolated lakes. Scores were generally higher in lakes I1 and I2 than I3. In Lake I1, axis 1 scores were lower during 2007 than 2005 and 2006 in spring and early summer although increased rapidly during the autumn and winter, which may reflect the pennate diatoms and *Asterionella* sp. observed in phytoplankton samples in spring, and the large biovolume of *Trachelomonas* sp. observed between October and December 2007. Axis 1 scores fell from January to March 2008 as a result of the increase in the biovolume of *Asterionella* sp. In Lake I2, axis 1 scores were generally lower during summer and then increased during autumn and winter during 2005 and 2006. In 2007, axis 1 scores did not increase as much, reflecting the abundance of *Cryptomonas* and *Rhodomonas* spp. and *Phacus* sp during winter, except for February 2008 when *Trachelomonas* sp. was relatively abundant. Axis 1 scores in Lake I3 showed a similar seasonal pattern as in connected lakes. In spring, Lake I3 axis 1 scores were lower than during 2005 and 2006, reflecting the high biovolumes of centric diatoms and *Aulacoseira* spp. During summer and winter, sample scores were similar on axis 1 during all three years.

Although phytoplankton axis 2 scores in the isolated lakes were largely homogenous between years, some samples had distinctly different scores. In Lake I1, phytoplankton axis 2 scores were low during late summer 2007 due to a combination of comparatively high biovolumes of *Dinobryon* sp. and *Trachelomonas* sp., and scores increased during February and March 2008 due to the phytoplankton community being dominated by *Asterionella* sp., *Mallomonas* sp. and centric diatoms. In Lake I2, phytoplankton axis 2 scores were high during the autumn and winter of 2007, particularly March 2008, when *Gymnodinium* sp., *Mallomonas* sp. and green flagellates became abundant. Lake I3 phytoplankton axis 2 scores were not substantially different during 2007 than the other years of the monitoring period. The high phytoplankton axis 2 score during spring 2005 was associated with a high abundance of green flagellates.

5.7.2 Zooplankton

Zooplankton abundance

Zooplankton community composition was compared between lakes in section 3.8 and the CA plot shown in Figure 3.52 (page 183). In connected lakes, zooplankton abundance typically increased during spring and summer although there was interannual and between-lake variability in the timing and extent of the increase in abundance (Figure 5.34). In Lake C1, zooplankton biomass generally increased during the spring to reach maximum abundances during summer then declined throughout autumn and remained low during the winter. The number of individual zooplankton was similar between 2005 and 2007, although a brief decrease was observed in total zooplankton individuals between June and July 2007. The maximum number of individual zooplankton occurred during October in 2005, September 2006 and August 2007. In Lake C2, the total number of zooplankton was similar between 2005 and 2006. During 2007, a brief increase in the total zooplankton population was observed to 13 ind. L^{-1} during March 2007, and then during the summer total zooplankton populations were similar during 2005 and 2006. Declines were observed in the total number of zooplankton between summer and winter in all years, after which less than 10 ind. L^{-1} were observed for the remainder of the year. During early summer 2007 in Lake C3 the total zooplankton population was similar to 2005 and 2006, although a large peak was observed during June. Summer zooplankton populations during 2007 were smaller than in either 2005 or 2006. In all three years, the total zooplankton population declined between October and November and remained low (<5 ind. L^{-1}) during the remainder of each year.

In isolated lakes, patterns of zooplankton abundance varied between lakes and between each year. In lake I1, the abundance of zooplankton did not exceed 10 ind. L^{-1} for the majority of 2005 and 2007. In both years, slight increases in the total abundance of zooplankton were observed during spring and late summer, and the abundance of zooplankton recorded each year were not substantially different. In Lake I2, total zooplankton abundance was similar between 2005 and 2006. In both years, zooplankton abundance increased during spring, and after decreasing during the early summer, increased throughout the summer. In 2006 the maximum zooplankton abundance occurred in September and was substantially higher than that observed

during 2005. The greatest total zooplankton abundance during 2007 occurred in August (35 ind. L^{-1}), early than either 2005 or 2006. In Lake I3, the highest abundances of zooplankton occurred during 2007. Both spring and late summer maxima were greater than those in either 2005 or 2006. In all three years, the highest zooplankton abundances were observed in April (2005 and 2007) or May (2006). During summer, zooplankton abundance was greatest between September and October in all years, although during 2007 was approximately twice that of either 2005 or 2006.

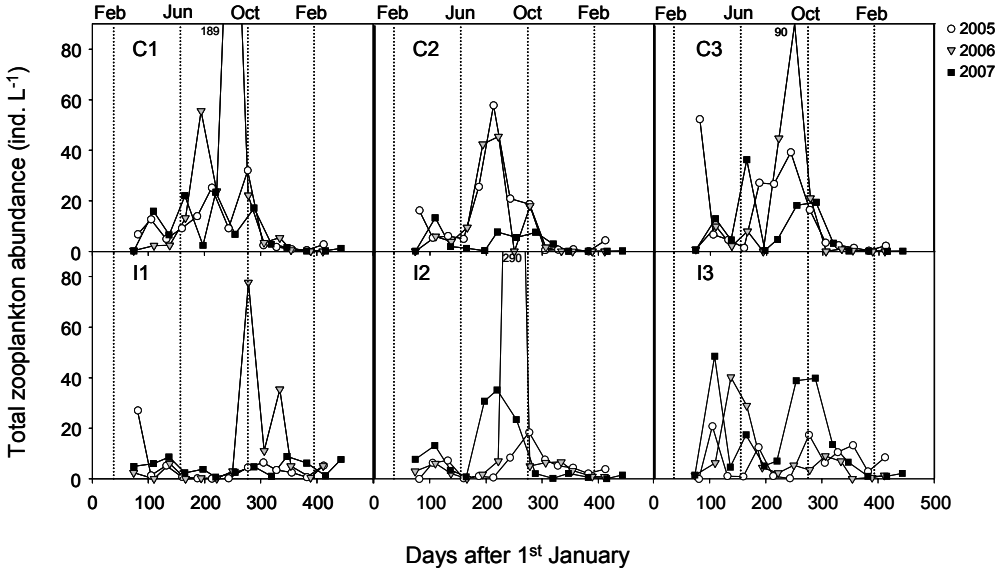

Figure 5.34 Total zooplankton abundance (ind. L^{-1}) plotted as numbers of days after 1st January for each year and each lake. 'Year' refers to March to 28th February (31st March for 2007).

Zooplankton community structure

Amongst connected lakes, axis 1 zooplankton sample scores did not vary substantially on a seasonal basis (Figure 5.35a). Samples from 2007 tended to have higher axis 1 scores during August than 2005 and 2006 due to an increase in the abundance of rotifers, although this change was stronger in Lake C1 than other connected lakes. Axis 1 scores in all three connected lakes decreased during the winter of 2006, because of the higher abundance of *Bosmina* spp. and *Ceriodaphnia* spp.

Zooplankton axis 2 sample scores (Figure 5.35b) in connected lakes were generally higher in the spring due to the abundance of *D. hyalina*. Axis 2 scores then tended to decrease during the summer as cyclopoid copepods became increasingly abundant. Axis 2 scores in Lake C1 increased sharply during summer 2007, when a large proportion of the zooplankton community comprised rotifers. During 2006, axis 2 scores were higher during winter than in 2005 and 2007, particularly in lakes C2 and C3. This may be a reflection of the presence of *Daphnia* in the winter zooplankton community.

Variability among years was evident in axis 1 scores in isolated lakes (Figure 5.35a). In lakes I1 and I2, axis 1 scores decreased during summer, indicating that *Daphnia* spp. were less abundant. Sample scores during 2006 and 2007 then increased and remained relatively constant for the remainder of the monitoring period. During 2005, sample scores remained lower than 2006 and 2007 in Lake I1, reflecting the comparatively low number of rotifers observed.

Sample scores in Lake I2 also decreased during the summer of all years of the monitoring period. During the autumn and winter of 2006 and 2007 axis 1 scores were lower than those observed during 2005 in Lake I2 as a result of the higher abundance of calanoid copepods during 2006 and 2007. Axis 1 scores in Lake I3 were generally higher than in other isolated lakes, particularly during summer 2005 when the rotifer *Keratella* sp. was present, although in winter 2006 axis 1 scores were low, reflecting the occurrence of *D. cucullata*.

Axis 2 scores were considerably different amongst isolated lakes (Figure 5.35b). In Lake I1, no similarity was observed between the axis 2 scores from each year, although during 2005 axis 2 scores were generally higher during the summer and winter, possibly reflecting the higher abundance of calanoid copepods during 2005. In Lake I2, axis 2 scores fell sharply during the summer in all years, suggesting that summer samples were associated with *Bosmina* sp. Sample scores for 2007 were not substantially different from those of 2005 and 2006. Axis 2 scores in Lake I3 were variable between years. A common feature was a decrease in scores during the winter of each year, associated with the presence of *D. cucullata* in the zooplankton

community. Samples from 2007 were not clearly different to those from either 2005 or 2006.

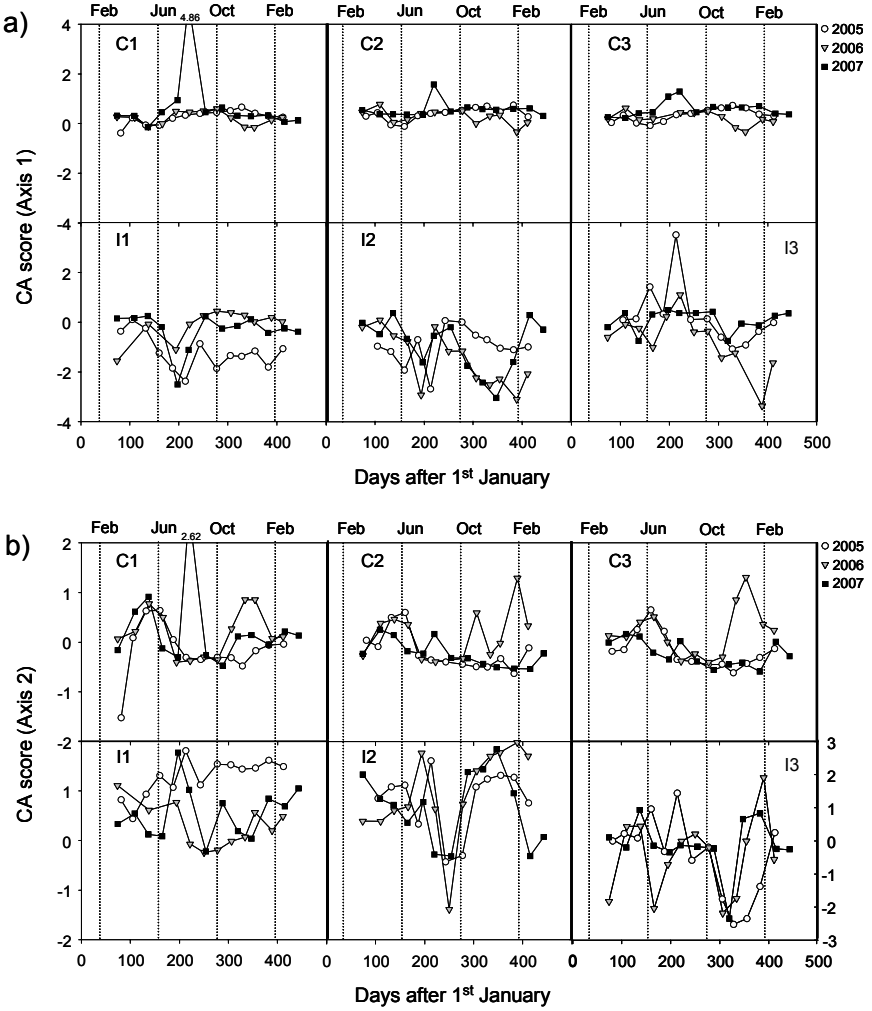

Figure 5.35 CA zooplankton sample scores for a) Axis 1, and b) Axis 2. Scores plotted as numbers of days after 1st January for each year and each lake. 'Year' refers to March to February (31st March for 2007). The species plot can be found in Figure 3.52 on page 183. Note the change in *y*-axis scores for Lake I3 in part b.

5.8 Discussion

5.8.1 The importance of rivers and streams as suppliers of nutrients

Inflowing rivers and streams to Attenborough Nature Reserve are important for supplying nutrients to the connected lakes and Lake I3. The significant differences observed between the inflow and in-lake concentrations of nutrients suggests that internal processes are important in determining in-lake nutrient concentrations. TP and SRP inflow concentrations were generally higher than in-lake concentrations during early spring (Figure 5.19a and b). During the spring, uptake of SRP by phytoplankton may be related to in-lake decreases in SRP concentration (Lampert and Sommer, 2007). Declines in SRP concentration during spring are a common feature in shallow lakes. Inflowing SRP may be particularly important in maintaining phytoplankton growth in the connected lakes and Lake I3 during this period. During the summer (particularly in 2005 and 2006), a shift towards internal loading appears to have occurred in the connected lakes, as in-lake P concentrations were similar to or higher than inflowing concentrations. This is likely to be a consequence of internal P loading. The release of P from the sediments of eutrophic lakes can be a highly important source of P during the late summer, which may exceed the loading from external sources (Søndergaard et al., 1999).

NH_4-N concentrations in inflowing water at Ea appeared to be more variable than for other nutrients (Figure 5.19c). Concentrations of NH_4-N were significantly higher in the Ea inflow water than the in-lake concentrations in the connected lakes. In-lake NH_4-N concentrations probably reflect a balance between the uptake of NH_4-N by phytoplankton and regeneration by zooplankton and breakdown of organic matter by bacteria. The difference between inflow and in-lake NH_4-N concentrations in connected lakes were greatest during the late summer when phytoplankton uptake is likely to be highest, suggesting the importance of external NH_4-N sources for phytoplankton growth. During spring, in-lake processes may have had a greater effect on NH_4-N concentrations. Zooplankton grazing appeared to be related to clear-water phases during the spring in the connected lakes (Chapter 4), which suggests that NH_4-N excretion by zooplankton may have been higher during spring. In Lake I3, NH_4-

N concentrations did not differ significantly between the inflow at BPI and in the lake. The small discharge of BPI suggests that internal processes are likely to have been most important for determining in-lake NH_4-N concentration. NO_3-N concentrations in the Eb inflow were significantly higher than in-lake concentrations, which suggests that the River Erewash is an important source of NO_3-N for connected lakes. Uptake of NO_3-N by phytoplankton is likely to be important in determining in-lake NO_3-N concentrations in the connected lakes and Lake I3, although other processes may also have an effect. The denitrification of NH_4-N to N_2 (Wetzel, 1983) and the reduction of NO_3-N to NH_4-N in anoxic conditions (Lampert and Sommer, 2007) may contribute towards the considerably higher inflow concentrations of NO_3-N compared to the in-lake observations in the connected lakes. The lower concentrations of NO_3-N measured in the MPO outflow compared to the Ea outflow suggest that the residence time of the water has a strong influence on chemical transformations that affect NO_3-N concentrations, for example by allowing more time for ammonification and denitrification (Saunders and Kalff, 2001b) and uptake by phytoplankton.

SiO_3 concentrations at Eb were significantly higher than the in-lake concentrations during the monitoring period (Figure 5.19e). Internal processes are also likely to strongly affect in-lake SiO_3 concentrations. The difference between inflow and in-lake concentrations was greatest during the spring and autumn when in-lake concentrations were substantially lower. Diatoms were abundant in connected lakes and in Lake I3, so the uptake of Si by diatoms was probably responsible for these decreases in SiO_3 concentration (Wetzel, 1983; Gibson et al., 2000). Release of SiO_3 from the sediments was probably important in both connected lakes and Lake I3. Since SiO_3 concentrations appeared to be almost entirely replenished after the growth of diatoms, internal recycling of SiO_3 may been an important additional supply of SiO_3 to the water-column in addition to external sources. Gibson et al. (2000) suggest that internal recycling of SiO_3 can play a significant role in determining in-lake concentrations in lakes with a residence time of more than one year, although internal recycling may have been important in the connected lakes which had short WRTs (<25 days).

The concentration of major ions in the inflows to the connected lakes and Lake I3 and observed in-lake concentrations were less seasonal than in lakes I1 and I2 (Figure

5.20). Major ions, such as K^+, Na^+ and Mg^{2+} are relatively conservative and so their concentration is not strongly mediated biologically, suggesting that in-lake concentrations of ions are primarily dependant on allochthonous processes and river in-flow (Krivtsov and Sigee, 2005), in contrast the uptake and release of major nutrients. The lower concentrations of major ions in isolated lakes I1 and I2 compared to connected lakes and Lake I3 also supports the importance of inflows as a suppliers of major ions. The concentration of major ions was frequently found to be higher than the in-lake concentrations which suggests that major ions may have sedimented out of the water column (Wetzel, 1983; Sigg, 1987).

The development of mass-balance nutrient budgets is frequently used to aid understanding of nutrient budgets in shallow lakes (e.g. Bengtsson, 1975; Nõges et al., 1998; Kozerski et al., 1999). The nutrient budgets developed for Attenborough Nature Reserve (Figure 5.22) suggest that the intermittent outflows from the connected lakes may have led to the retention of nutrients in the connected lakes during 2006.

Contrary to expectations, the nutrient budgets did not consistently indicate release of P during the summer (Bengtsson, 1975; Søndergaard et al., 1999; Spears et al., 2006). However, this is more a reflection of discharge to the connected lakes being greater than the outflows. It is likely that P release from the sediments did occur, as suggested by the higher concentrations of TP during the summers of 2005 and 2006. Much of the P released from the sediments during summer is likely to have been retained in the connected lakes due to the low outflow. 2007 was the only year during which P released from the sediments was diluted and flushed from the connected lakes.

The finding that the connected lakes appeared to be a sink of NO_3-N during 2005 and 2006 is in line with the findings of Saunders and Kalff (2001b) that lakes are sinks of N, although assessing the relative effects of NO_3-N losses from reduction to NH_4-N and the lack of substantial outflow during the summer of 2005 and 2006 on this calculation is difficult. During 2007 large quantities NO_3-N were lost from the connected lakes, which is probably mainly associated with the high discharge at MPO. NH_4-N losses were also greatest during 2007, and was retained during 2006.

Si has been observed to be released during the autumn and winter in other shallow lakes (e.g. Gibson et al., 2000) although this was not apparent from the nutrient budget here. Estimates suggest that more SiO_3 was retained in the connected lakes during 2006 than during 2005. Sedimentary release is likely to have been relatively less significant compared to inflow as in-lake concentrations were nearly always less than inflow concentrations (Figure 5.19).

It appears that WRT exerts a strong influence on net nutrient retention in the connected lakes, although this may be a debateable conclusion since the spot-measurements made of outflow volume and nutrient concentration are likely to be subject to high errors in the measurement of nutrients leaving the lakes. This is particularly evident during 2007, when large outflow volumes were observed. Other studies have used higher frequency of sampling (e.g. daily, Nõges et al., 1998; weekly, Bengtsson, 1975; Kozerski et al., 1999) to provide more robust estimations of nutrient loss. Future studies should use higher resolution monitoring of outflows and their nutrient concentrations to enable more precise measurements of nutrient loss and retention.

5.8.2 Effects of hydrological change on chemistry and biology of connected and isolated lakes

This section discusses the changes in chemical and biological components that were observed during the very wet year of 2007. The changes in hydrology that occurred are summarised in Figure 5.11 (page 238) and section 5.3.6 (page 244). In connected lakes, WRT was reduced, MPO became the dominant outflow (Figure 5.10) and water depth increased. In isolated lakes water depth increased but there was no quantifiable change in WRT. In summary, the key chemical changes observed in the connected lakes were a reduction in summertime P concentrations probably because of dilution and flushing, and dilution of the increased loading of other major nutrients. Total phytoplankton biomass was reduced and cryptophytes and diatoms became the dominant phytoplankton groups. Total zooplankton biomass declined during summer 2007. In isolated lakes, dilution associated with increased water depth decreased TP

and chlorophyll-*a* concentrations, and reduced the abundance of cyanobacteria and chlorophytes during 2007.

Chemical response

In-lake TP concentrations in the connected lakes were reduced during the wet year of 2007. This may be related to the dilution and washout of the summertime P release. The factors which affect P release from sediments for which measurements were made (e.g. temperature, pH, nitrate concentrations; Søndergaard et al., 2003) were not substantially different during 2007 compared to 2005 and 2006. Other mechanisms may affect P release (e.g. redox conditions, resuspension, bioturbation; Søndergaard et al., 2003) were not directly measured, although it is unlikely that they would have been significantly different compared to the previous two years. Therefore, P release is still likely to have occurred, but washout and dilution of the released P reduced in-lake P concentrations. Elliott et al. (2009) suggested that the in-lake concentration of nutrients derived entirely from a point-source would be reduced because of dilution by increased river discharge. These findings support this as the concentration of TP and SRP was negatively correlated with the discharge of the River Erewash (Figure 5.36). Cooler water temperatures during 2007 may also have reduced P release (Jensen and Andersen, 1992).

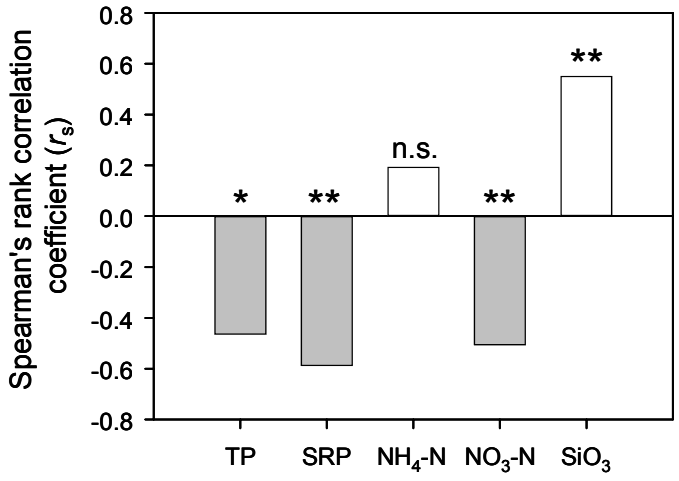

Figure 5.36 Summary of Spearman's rank correlation coefficients of monthly inflowing nutrient concentration at Eb with River Erewash monthly discharge. *, $p \leq 0.05$; **, $p \leq 0.001$; n.s, not significant.

No correlation between River Erewash discharge and NH_4-N concentration at Eb was found. NH_4-N concentration in the inflowing water is likely to be affected by the treatment of sewage in STWs and anoxic conditions (which may increase NH_4-N regeneration, e.g. Beutel, 2006). Few published studies have found clear trends in the response of in-lake NH_4-N concentrations to flood events. Paidere et al. (2007) and Glińska-Lewczuk (2009) found that increased hydrological loading increased in-lake NH_4-N concentrations, although Roozen et al. (2008) have shown that flood events reduced NH_4-N concentrations in Dutch floodplain lakes due to dilution.

The concentration of NO_3-N in inflowing water was negatively correlated with the monthly discharge of the River Erewash (Figure 5.36). Since the in-lake concentrations of NO_3-N did not change substantially in response to the increased loading during 2007, it is likely that at high discharge, the increased transport of NO_3-N into the lakes was sufficient to prevent changes in in–lake concentration. Other authors (e.g. Paidere et al., 2007; Roozen et al., 2008; Mihaljević et al., 2009) have observed increased in-lake concentrations of NO_3-N as a result of flooding events, although these studies were mainly concerned with more rural catchments where diffuse nutrient loading is likely to be higher.

SiO_3 concentrations were positively correlated with discharge (Figure 5.36). Si in rivers is largely derived from diffuse sources (Sferratore et al., 2006), and so loading of SiO_3 increases with river discharge. More diffuse-source nutrients are supplied to lake basins when discharge increases, although simultaneously are removed from the lake by increases in flushing rate (Elliott et al., 2009). This may account for the lack of increase in SiO_3 concentrations during 2007 despite the increase in SiO_3 loading. Few published studies have specifically considered the effects of flooding on SiO_3 concentrations in shallow lakes, although both Paidere et al. (2007) and Roozen et al. (2008) found increases in SiO_3 concentrations as a result of flooding. However, both studies considered the impact of flood-pulses rather than changes in hydraulic loading.

Biological response

During 2007, chlorophyll-*a* concentration was less than during 2005 and 2006 (see section 5.7.1). This was observed in both connected and isolated lakes. Connected

lake maximum chlorophyll-*a* concentration was around 50% less than during 'normal' years, and in isolated lakes I1 and I2 the proportional reduction was substantially greater. In the connected lakes, this is most likely to have been the result of simultaneous increases in dilution and washout. The absence of significant outflows from lakes I1 and I2 suggests dilution related to increases in lake volume is a likely cause in the isolated lakes. Estimates suggest that lake volume increased by approximately 25% in Lake I1 and 36% in Lake I2 during the summer of 2007. Several authors have found reductions in total phytoplankton abundance associated with increased flushing rate (e.g. Brook and Woodward, 1956; Jones and Elliot, 2007; Paidere et al., 2007; Haldna et al., 2008). Dickman (1969) found that phytoplankton biomass increased and larger species became more dominant in an isolated section of a well flushed lake. Some evidence suggests that the timing of wet periods alters the effects of phytoplankton biomass response. Large flooding events in floodplain lakes of the River Danube during March and June delayed the occurrence of maximum phytoplankton biomass until September (Mihaljević et al., 2009). During a dry year, maximum phytoplankton abundance occurred during June. A similar delay was observed in connected lakes and isolated lakes I1 and I2 during 2007, when following an early summer wet period, maximum phytoplankton biomass did not occur until September to October.

The phytoplankton communities of both isolated and connected lakes during 2007 was associated with rapidly reproducing taxa. Diatoms and cryptophytes were relatively abundant, whilst cyanobacteria and chlorophytes were rare compared to the previous summers. The dominance of *Cryptomonas* sp. during flood events has also been observed by García de Emiliani (1997), Huszar and Reynolds (1997) and Paidere et al. (2007) in floodplain lakes. Diatoms appear to be resistant to flood events. Huszar and Reynolds (1997) reported that the diatom *Cyclotella* sp. was abundant during flooding and Paidere et al. (2007) observed *Synedra* sp., *Nitzschia* sp. and *Navicula* sp. to be dominant taxa during summer flood events. The reasons for the selection of cryptophytes and diatoms over cyanobacteria and chlorophytes during wet years may be associated with the ability of both cryptophytes and diatoms to reproduce relatively quickly (Reynolds, 2006), which could help them to compensate for losses by dilution and washout more rapidly than some other genera. Diatoms may also have been favoured by high Si loading during wet years. Applying Grime's (1977) theoretical *C*-

S- *R*- strategy paradigm to phytoplankton (see Table 1.7), suggests that *R* strategists, which includes some diatoms, could be favoured in highly disturbed environments. The low light requirement of *R* strategists may also be a competitive advantage (Reynolds, 2006) when minerogenic turbidity associated with flooding is high.

Differences in total zooplankton abundance were not widely observed during 2007 (see section 5.7.2) compared with 2005 or 2006. In isolated lakes I1 and I2 zooplankton abundance was similar to that observed during 2005, although summer maxima in 2006 were substantially greater. A lack of response in isolated lakes suggests that zooplankton were not affected by the increase in water depth. In connected lakes, reductions in zooplankton abundance coincided with the June to July flood event in 2007. Brook and Woodward (1956) observed that zooplankton abundance was greatest in Scottish lakes with longer residence times. In the Neuendorfer See, Germany, Walz and Welker (1998) found that zooplankton abundance decreased as WRT decreased. The effect was particularly strong when WRT was < 8 $days^{-1}$. Angeler et al. (2000) found that a decrease in zooplankton biomass during more lotic periods resulted in a reduction in the strength of top-down effects on phytoplankton in floodplain lakes in Spain. However, zooplankton community structure at Attenborough Nature Reserve was largely unresponsive to changes in hydrology. Axis scores did not vary substantially between wet and dry years, implying that hydrological changes did not alter the zooplankton community. Other studies including the response of zooplankton to flood events have found that rotifers become more abundant because of their short regeneration times (e.g. Walz and Welker, 1998; Angeler et al., 2000; Paidere et al., 2007). The reduced zooplankton abundance observed particularly in lakes C2 and C3 suggests that a reduction in the strength of top-down processes occurred during 2007 (Angeler et al., 2000). A side effect of the isolation of connected lakes from the River Erewash may be an increase in zooplankton abundance and a strengthening of top-down effects.

5.8.3 Potential effects of future hydrological change

As discussed in section 5.5.2, a Vollenweider model was used to investigate how future hydrological changes may affect the mean annual in-lake TP concentrations in the connected lakes (page 256). The model suggested that diverting the River

Erewash may have little effect on the mean in-lake TP concentration of connected lakes. The degree to which the model can predict future change is strongly dependent on its ability to accurately model observed data. Modelled TP concentrations in the connected lakes were reasonably similar to mean measured in-lake TP concentrations (Figure 5.23). However, the application of basic Vollenweider models to shallow lakes is often criticised, as they do not model P release from the sediments during the summer, which in eutrophic shallow lakes can account for substantial a proportion of in-lake TP concentration (Søndergaard et al., 1999, 2003). This may account for the improved performance of the model during summer 2007 when P release was diluted by the high inflow. The reasonable correspondence between measured and modelled TP concentrations may reflect the short WRT of the connected lakes, particularly during 2007, as the effect of internal processes (such as P release from the sediments) is reduced with decreasing WRT (Scheffer, 1998).

Mean annual chlorophyll-*a* concentrations were always overestimated by the Vollenweider model (Figure 5.24). A number of potential issues have been identified with the interpretation of predicted chlorophyll-*a* values from Vollenweider models, even if they are 'not the fault of the model [but] relate to its simplistic interpretation' (Reynolds and Davies, 2001). In this case, the interpretation of the models output should acknowledge three shortcomings. Firstly, grazing by zooplankton, particularly during the spring clear-water phase, may contribute to overestimations of mean chlorophyll-*a* concentration, which may be particularly important in the connected lakes due to the strong top-down effects of zooplankton grazing discussed in Chapter 4. Secondly, self-shading, associated with the high biomass of phytoplankton (Reynolds, 1984), and minerogenic turbidity, derived from the River Erewash, may also limit the realised phytoplankton abundance (Donohue and Molinos, in press). Thirdly, the model does not simulate washout of phytoplankton (Elliott et al., 2009) which may account for the particular overestimate of mean chlorophyll-*a* concentrations for 2007. Therefore, direct estimates of chlorophyll-*a* concentration in future hydrological and nutrient loading scenarios must be made with caution, but can provide insight into the potential maximum annual chlorophyll-*a* concentration.

These results have important management implications for Attenborough Nature Reserve. Dilution of in-lake TP concentrations has been advocated as a potential

restoration technique for other shallow lakes. If river discharge into lakes is restricted to the summertime, in-lake P concentrations can be reduced by the flushing of P released from the sediments (Spears et al., 2006). Summer 2007 fortuitously showed the potential merits of this approach in Attenborough Nature Reserve. Due to the existing weirs this technique may be a promising option for reducing in-lake TP concentrations. However, hydrological modelling undertaken as part of the River Erewash diversion project suggests that the quantity of water entering the connected lakes from storms (when discharge exceeds 1.5 m^3 sec^{-1}) during the summer (1 May to 30 September) will be 6% of total summer storm discharge in the River Erewash (HR Wallingford, 2006). This is likely to prevent the flushing of internally-loaded P which will accumulate in the water column and may lead to negligible reductions of in-lake P concentrations. Data also suggest that the River Erewash supplies N and Si to the lakes. Reducing the supply of N and Si, whilst P remains abundant, may potentially favour the growth of N-fixing cyanobacteria (Krivtsov et al., 2000a) at the expense of diatoms and chlorophytes, which would present a significant management challenge to Attenborough Nature Reserve. However, the model predicted comparatively large decreases in in-lake TP concentrations (Figure 5.24a) as a result of reducing inflowing nutrient concentrations, which suggests that restoration measures should be targeted towards improving P removal at STW and maintaining the current hydrological connectivity.

5.9 Conclusion

This chapter has demonstrated the importance of integrating hydrology into the monitoring of Attenborough Nature Reserve in order to ensure successful management. The chapter has shown that inflowing rivers are important suppliers of nutrients to lakes into the reserve, as evident in the high concentrations of major nutrients in the River Erewash and in the inflow stream to Lake I3. This finding suggests that measures to reduce nutrient concentrations in Attenborough Nature Reserve must initially focus on inflowing rivers and streams, before internal measures are implemented. The development of nutrient budgets suggests that the nutrient loading to the connected lakes is dependent on the discharge of the River Erewash

and internal loading. High-resolution hydrological data are important for the assessment of nutrient retention and release.

The Attenborough Nature Reserve is a complex hydrological system. The water level in the connected lakes can be affected by a combination of the discharge of the Rivers Erewash and Trent. Isolated lakes appear to respond principally to groundwater inputs. The unusually wet year of 2007 enabled a comparison of hydrological functioning and chemical and biological response. The route of flows through the connected lakes during summer 2007 was altered as a result of the blocking of outflows at Ea because of the increase in discharge in the River Trent, and caused change to a more lotic system with a very short WRT. These conditions may be responsible for three changes observed in lake ecology and chemistry. 1) P concentrations were reduced during 2007, particularly during the summer, suggesting that a combination of dilution and flushing suppressed the internal loading. A lack of change in the concentration of other nutrients appears to be a result of dilution of the increased nutrient loading. 2) Total phytoplankton biomass was reduced, with cryptophytes and diatoms becoming dominant. 3) Total zooplankton biomass also declined. In isolated lakes, reductions in TP and total phytoplankton biomass also occurred with cyanobacteria and chlorophytes becoming more scarce.

The application of a Vollenweider model to the connected lakes suggests that in-lake TP concentrations are primarily determined by the concentration of inflowing nutrients. Modelled TP concentrations did not respond much to reductions in River Erewash discharge. Although total isolation would prevent the input of nutrients into the connected lakes, a reduction in flushing rate increases the strength of in-lake processes, such as P release from the sediments. Since the supply of N and Si is also likely to be reduced by the diversion of the River Erewash, the growth of nitrogen-fixing and possibly bloom-forming cyanobacteria may be favoured. This suggests that the use of river diversion to reduce the in-lake nutrient concentration of the connected lakes of Attenborough Nature Reserve may not be an effective solution and may lead to the maintenance of high P concentrations and aesthetically unacceptable cyanobacteria blooms.

A potentially effective method for the restoration of the connected lakes could be reducing the concentration of P in inflowing water. Maintaining hydrological connectivity to the River Erewash could potentially allow dilution during summer storms. Maintaining a high flushing rate might also help in preventing the growth of bloom-forming cyanobacteria. Existing weir structures could be used in order to allow rapid outflow, and therefore increase the flushing rate, during summer storms to aid the washout of phytoplankton and nutrients.

Chapter 6 Experimental effects of isolation and N and Si manipulation on lake plankton

6.1 Introduction

It is well established that the concentration of P is the principal determinant of the maximum biomass of phytoplankton that can be supported by a lake (Dillon and Rigler, 1974). This has meant that lake restoration has typically been principally focussed on reducing P concentrations in order to reduce chlorophyll-*a* concentrations (e.g. Bootsma et al., 1999; Madgwick, 1999; Rip et al., 2005). Changes in the concentrations of other nutrients, such as N and Si, may occur when reducing the P loading to lakes (Søndergaard et al., 2005). Chapter 5 suggested that changes in P, N and Si ratios may occur in lakes connected to nutrient-rich inflows during flood events. Both N and Si may play a role in determining the total biomass and composition of phytoplankton communities in eutrophic lakes, although the extent of these changes and the mechanisms by which they may occur are not well understood in shallow lakes.

N may be particularly important in highly eutrophic environments where denitrification rates can be high, as phytoplankton communities may be limited by N during the summer (section 1.4.2, Wetzel, 1983; Petzoldt and Uhlmann, 2006). Data from Attenborough Nature Reserve suggest that in the Erewash-connected lakes, N is strongly depleted during the summer months (Chapter 3 and Chapter 4). Low N concentrations during the summer in isolated lakes I1 and I2 suggests that N-fixing cyanobacteria may be favoured (Schindler, 1978; Smith, 1983). Cyanobacteria are aesthetically unpleasant and may pose a danger to the health of humans and livestock (Scheffer, 1998). The diversion of the River Erewash may remove an important source of N from the connected lakes and increase the likelihood of cyanobacteria becoming dominant. Therefore, the effects of N on shallow lake phytoplankton may be of importance for lake management, particularly when reducing P loading.

Si may also play a role in determining phytoplankton community structure (section 1.4.3). Si is particularly important for the growth of diatoms, and in eutrophic shallow lakes Si can become limiting for diatom growth (e.g. Gibson et al., 2000). SiO_3 concentrations in the connected lakes and Lake I3 of the Attenborough Nature Reserve were frequently depleted concentrations that may be limiting (~0.5 mg L^{-1}, e.g. Figure 3.2d, page 94) and therefore diatom productivity may have been dependent on the Si supply from the River Erewash. Reductions in Si concentrations have been simultaneously observed with P reduction measures (Phillips et al., 2005; Søndergaard et al., 2005). Si limitation has been associated with a switch from diatom to cyanobacteria dominance in eutrophic lakes (e.g. Horn and Uhlmann, 1995; Krivtsov et al., 2000a) and at Attenborough Nature Reserve, diatoms were rare in lakes without an external supply of Si (Lake I1, Figure 3.29, page 139; Lake I2, Figure 3.37, page 152). Therefore, understanding the effects of Si concentrations on phytoplankton may be advantageous for the management of Attenborough Nature Reserve.

River diversion will also substantially alter the hydrological regime of the lakes. Evidence presented in Chapter 5 implied that high flushing rates reduced the abundance of cyanobacteria and also reduced in-lake P concentration. The discussion in section 3.9 (page 187). Chapter 3 suggested that internal release of P may be important in determining late summer phytoplankton biomass in the connected lakes. Isolation of the connected lakes by river diversion may therefore favour slower-growing cyanobacteria, as appears to occur in isolated lakes I1 and I2, which could represent a significant management challenge for Attenborough Nature Reserve. Additionally, P may accumulate in the water column when flushing is low by preventing the washout of P released from the sediments (section 5.6.1). However, because the diversion of the River Erewash will probably increase the importance of groundwater and rainfall as sources of water to the currently connected lakes, there may be some dilution of nutrients. Resolving the potential effects of river diversion is important for the management of Attenborough Nature Reserve.

This chapter therefore aims to use a mesocosm experiment for two purposes: i) to simulate the effects of river diversion on lake plankton and water chemistry, and ii) to

investigate the main and interactive effects of N and Si on plankton and water chemistry. The unique nature of this experiment is in the transfer of relatively low-nutrient concentration water from Lake I2 into mesocosms built in Lake C2, which does not appear to have been attempted in previous studies at this scale. Monitoring data showed that P, N and Si concentrations were lower in Lake I2 (see section 3.6.1, page 146) than in Lake C2 (section 3.3.1, page 106) during the summer. The transfer of lower nutrient concentration water was designed to simulate the effect of reducing P loading to Lake C2, and allowed N and Si concentrations to be manipulated individually, including a reduction in concentration. The design of the mesocosms maintained contact between the isolated water column and the sediments, to ensure that interactions between the water column and the sediment could take place during the experiment, simulating the effects of isolating the connected lakes from the River Erewash. Specifically, the following hypotheses were tested:

- Simulated lake isolation reduces the total phytoplankton biomass and the concentration of nutrients,
- N addition increases the growth of phytoplankton biomass and reduces the abundance of cyanobacteria,
- Si addition increases the abundance of diatoms in the phytoplankton.

6.2 Methods

Three stages were required to build the mesocosms and apply the nutrient manipulations. These were (i), the design and construction of the mesocosms; (ii), replacing the water in the mesocosms; and (iii), manipulating the concentrations of NO_3-N and SiO_3 in the mesocosms. Sampling procedures and statistical techniques are given in sections 6.2.4 and 6.2.5.

6.2.1 Mesocosm design and construction

The construction of the mesocosms took place in Lake C2 during July 2007. The location of the mesocosms (Figure 6.1) was determined largely by the requirement for

soft sediments for ease of construction, suitable water depth and proximity to Lake I2 to allow the transfer of water. The water in the open-ended mesocosms was in contact with the sediments of Lake C2 (Figure 6.2a), in order to maintain the interaction between sediments and the water column. The mesocosms were constructed in water approximately 1.5 m deep. The diameter of the mesocosms was 1 m, with an area of 0.78 m^2. Each mesocosm held approximately 1200 L of water.

Four frames constructed of wood were used to support the mesocosms (Figure 6.2a and b). Fence posts were firmly hammered into the lake sediments to anchor the frame, which was then lowered over the top of the fence posts and attached to the top of the posts (Figure 6.3a). Flexible translucent polythene tubing, approximately 250 µm thick, was used to form the walls of the mesocosms. (Figure 6.3b). The top and bottom of the tubing was attached to circles of plastic tubing and fixed using cable-ties. At the bottom, weights were attached to the tube circles and firmly anchored into the sediments. The top of each tube was then fixed to the wooden frame in the lake. Due to the unexpected increases in water depth associated with the flooding observed during the summer of 2007 it was necessary to increase the height of the mesocosms after floodwater caused overtopping between the mesocosms and Lake C2 before the transfer of water (Figure 6.3c). Fortunately this occurred prior to the transfer of water and start of the experiment. After the increase in height, the tops of the tubes were at least 30 cm above the surface of the water during the experiment (Figure 6.3d). By the end of the experiment, the water surface was around 60 cm below the top of the tubes.

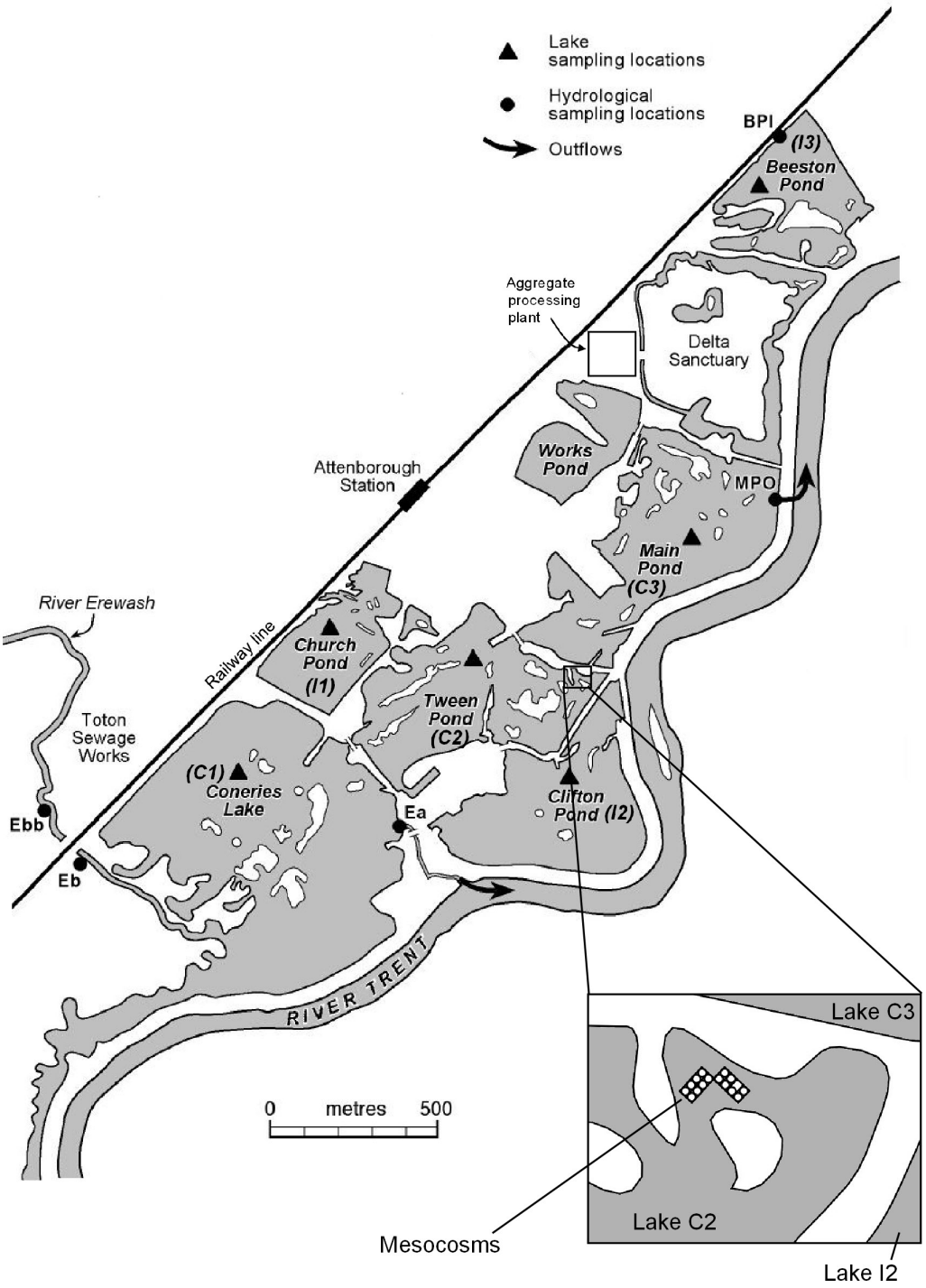

Figure 6.1 The location of the mesocosms within Attenborough Nature Reserve.

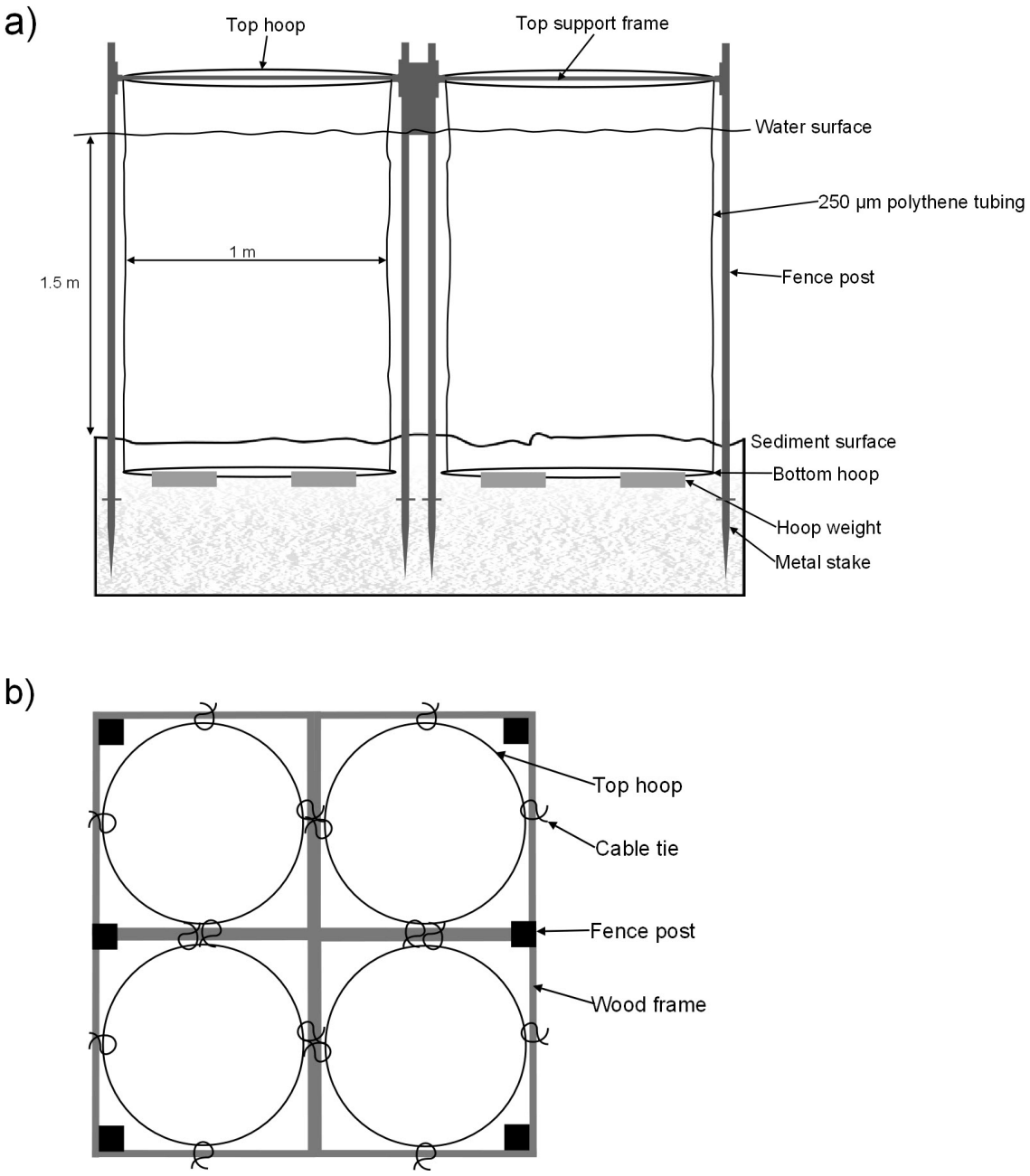

Figure 6.2 The design of the mesocosms used during the experiment: a) elevation view; b) plan view.

a)

b)

c)

Figure 6.3 Construction of the mesocosms: a) wooden frames attached to fence posts hammered into the lake sediments; b) mesocosm walls being made from polythene tubing attached to hoops of plastic pipe; c) flooding before the experiment causing overtopping (foreground) and in the background, extensions added to existing frames.

d)

e)

f)

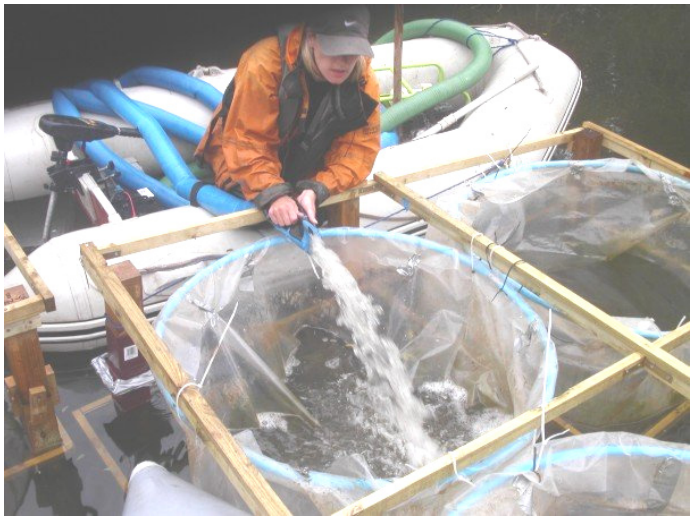

Figure 6.3 contd: d) completed mesocosms; e) removal of water inside mesocosms using pump and water butt immersed in mesocosm (not visible); f) refilling a control mesocosm with water from Lake C2.

6.2.2 Water replacement

Water was removed from all mesocosms and replaced using two commercial centrifugal water-pumps, each with a capacity to pump at approximately 600 L hr^{-1}. Water was pumped from the mesocosm and replaced with water from Lake I2 or C2 as appropriate. A weighted water-butt with holes drilled into the sides was immersed in the mesocosms and gently lowered to the sediment surface, with care taken not to avoid excessively disturbing the lake sediments. This was done to prevent the mesocosm collapsing as the water was removed (Figure 6.3e). Water from Lake I2 was obtained by pumping into large water-butts, which were transported on boats and then siphoned into the emptied mesocosms. Control mesocosms were emptied of water using the same technique and then re-filled using Lake C2 water (Figure 6.3f).

6.2.3 Experimental design

A randomised design with three replicates of each treatment and control was used. The treatments were: NO_3-N addition (+N), SiO_3 addition (+Si), NO_3-N and SiO_3 addition (+N+Si), water transfer from Lake I2 alone (-N-Si) and a control treatment (Ctrl) (Lake C2 water) (Table 6.1). The concentration of SiO_3 in the mesocosms was manipulated through the addition of sodium metasilicate (Na_2SiO_3), and calcium nitrate ($Ca(NO_3)_2$) was used to increase the NO_3-N concentration. The aim of the manipulations was to raise the concentrations of N to 10 mg L^{-1} and Si concentrations to 6 mg L^{-1}. These concentrations were chosen as they were similar to those observed in connected lakes during the winter months of the monitoring period (Figure 3.2, Figure 3.10, Figure 3.18), and so represent an estimate of NO_3-N and SiO_3 concentrations when the influence of uptake by phytoplankton would be lowest. Solutions of each nutrient were prepared in the laboratory using distilled water, and then the appropriate quantity added in the field. Quantities of concentrated nutrient additions were calculated based on analysis of the chemistry of samples taken 24 hrs previously and the volume of the water in the mesocosms. The initial addition of nutrients took place on 26 July 2007 (day 0), and additions were made as required every 7 days after the start of the experiment.

Table 6.1 Summary of water sources and the treatments applied to mesocosms.

Treatment	Source water	N concentration	Si concentration
+N	Lake I2	10 mg L^{-1}	Lake I2 ambient
+Si	Lake I2	Lake I2 ambient	6 mg L^{-1}
+N+Si	Lake I2	10 mg L^{-1}	6 mg L^{-1}
-N-Si	Lake I2	Lake I2 ambient	Lake I2 ambient
Control	Lake C2	Lake C2 ambient	Lake C2 ambient

6.2.4 Sampling procedures

The mesocosms were sampled six times, every 7 days after 26 July until 30 August. The collection and analysis of water, phytoplankton and zooplankton samples was carried out as described in Chapter 2, except that only the top 1.0m of water was sampled from the mesocosms using a Perspex tube, and measurements of dissolved oxygen, temperature, and specific conductivity were only made at a depth of 50 cm. The methods used for water chemistry are also given in Chapter 2. Water chemistry, phytoplankton and zooplankton samples from Lake C2 were taken simultaneously using the same techniques.

6.2.5 Statistical analysis

Two analyses were performed on the data obtained from the mesocosm experiment in order to investigate the two aspects of the experiment. Both approaches were based on repeated-measures analysis of variance (RM-ANOVA), as successive measurements were made over time on the same experimental units (mesocosms) (Everitt, 1995; Chan, 2004). The first approach was to simulate the effects of lake restoration, and the second to investigate the effects of N and Si on lake plankton. The effects of lake restoration were investigated using a one-way RM-ANOVA. Time was the one repeated factor and treatment (+N, +Si, +N+Si, -N-Si, and control, see Table 6.1) was the only fixed factor (Von Ende, 2001). A second RM-ANOVA was then used to test for the interaction between N and Si. The first RM-ANOVA could not analyse the interaction between N and Si because the design was not fully-factorial. By excluding the control treatment analysis, N addition and Si addition could each be used as fixed factors and the data reanalysed as a fully-factorial experiment as all

combinations of addition or non-addition of N and Si were present (Gravetter and Wallnau, 2000), allowing for the interaction between them (N × Si) to be analysed. Time was also used as the repeated factor in the second RM-ANOVA.

Where a visual examination of residuals suggested a significant departure from normality, $\log_{(x+1)}$ transformations were applied to the data to stabilise variances. If Mauchley's test indicated the assumption of sphericity had been violated, the Greenhouse-Geisser correction was applied to the degrees of freedom used with the *F* ratio (Field, 2000). Since the estimates of sphericity did not exceed 0.75, the Greenhouse-Geisser correction was considered most appropriate (Field, 2000). The Greenhouse-Geisser correction is more conservative than the Huynh-Feldt adjustment reduces and therefore reduces the likelihood of Type I errors (Von Ende, 2001). Homogeneity of variance in the ANOVA models was tested using Levene's test. Where significant ($p \leq 0.05$) differences in variance among treatments were detected, *post-hoc* comparisons were made with the Games-Howell test in the first RM-ANOVA. The Games-Howell test is considered a robust post-hoc test when sample variances are unequal (Field, 2000). Differences in treatment means at the start of the experiment (0 days) were tested for using one-way ANOVA with transformed datasets where appropriate.

Because RM-ANOVA requires no missing data, and no recognised procedure exists for dealing with missing data, missing values were assigned the mean values of other mesocosms of the same treatment on the same sampling day (e.g. Hopkins et al., 2002). Although this increases the degrees of freedom and increases the risk of a Type I error (Underwood, 1997), the number of missing data points was small (no more than two per variable for the duration of the experiment, except for total alkalinity where data was missing for one sampling date), and replacing the missing values allowed for a balanced RM-ANOVA to be used. All mean values given in the text have one standard error reported in parentheses. SPSS 15.0 was used for all univariate analyses.

Lake samples were compared to control samples to analyse the effects of isolation. Because there was no replicate of the lake sample, it was assumed that the error in the measurement of the control was equal to that error in the measurement of the lake

sample, in order to improve the robustness of the test. In order to estimate the significance of difference between the lake and control mesocosms, the standard error of the mean usually used in a one-sample *t*-test was substituted for the standard error of the difference between the control mesocosm and the lake. By estimating the error associated with the lake samples, this approach effectively created an independent-samples *t*-test.

Multivariate analysis of phytoplankton and zooplankton samples was performed using CANOCO 4.5 (ter Braak and Šmilauer, 2002). Genera which contributed less than 5% of the total phytoplankton biovolume of any sample were removed prior to analysis (e.g. Raikow et al., 2004). Both phytoplankton and zooplankton datasets were $\log_{(x+1)}$ transformed prior to analysis to stabilise variances (Ramette, 2007). Initially, a DCA was performed on phytoplankton and zooplankton datasets to assess the length of axis 1 in order to determine the most appropriate form of multivariate analysis. Since the DCAs of each dataset revealed that the length of axis 1 was less than 2 standard deviations, the linear ordination technique principal components analysis (PCA) was used (Jongman et al., 1987).

Table 6.2 Transformations applied to the datasets (and units of measurement in parentheses) used in the analysis of the mesocosm experiment.

Variable	Transform	Variable	Transform
Chemical		**Phytoplankton**	
TP (µg L^{-1})	none	Chlorophyll-*a* (µg L^{-1})	$\log_{(x+1)}$
SRP (µg L^{-1})	none	Bacillariophyceae (µm^3 mL^{-1})	$\log_{(x+1)}$
NO_3-N (mg L^{-1})	$\log_{(x+1)}$	Chlorophytes (µm^3 mL^{-1})	$\log_{(x+1)}$
NH_4-N (mg L^{-1})	$\log_{(x+1)}$	Cyanobacteria (µm^3 mL^{-1})	$\log_{(x+1)}$
SiO_3 (mg L^{-1})	none	Cryptophytes (µm^3 mL^{-1})	$\log_{(x+1)}$
Mg^{2+} (meq L^{-1})	none	Dinophytes (µm^3 mL^{-1})	$\log_{(x+1)}$
Ca^{2+} (meq L^{-1})	none	Euglenophytes (µm^3 mL^{-1})	$\log_{(x+1)}$
Na^{+} (meq L^{-1})	none	Chrysophytes (µm^3 mL^{-1})	$\log_{(x+1)}$
K^{+} (meq L^{-1})	none	**Zooplankton**	
Cl^{-} (meq L^{-1})	$\log_{(x+1)}$	Total zooplankton (ind. L^{-1})	$\log_{(x+1)}$
Total alkalinity (meq L^{-1})	$\log_{(x+1)}$	*Daphnia* spp. (ind. L^{-1})	$\log_{(x+1)}$
Physico-chemical		*Ceriodaphnia* spp. (ind. L^{-1})	$\log_{(x+1)}$
Temperature (°C)	none	Cyclopoid copepods (ind. L^{-1})	$\log_{(x+1)}$
Specific conductivity (ms cm^{-1})	none	Calanoid copepods (ind. L^{-1})	$\log_{(x+1)}$
Dissolved oxygen (mg L^{-1})	none	*Bosmina* spp. (ind. L^{-1})	$\log_{(x+1)}$
pH	none	Rotifers (ind. L^{-1})	$\log_{(x+1)}$
Secchi disk depth (cm)	none		

6.3 Results

6.3.1 Water chemistry and physico-chemical variables

Treatments

During the experiment, NO_3-N concentrations in Lake C2 did not vary substantially (Figure 6.5a), ranging between 1.9 and 2.6 mg L^{-1}. RM-ANOVA showed that NO_3-N concentrations were significantly higher in the +N and +N+Si treatments than in other treatments and in the control mesocosms (Table 6.3) during the experiment, suggesting that the manipulation of NO_3-N was successful. Decreases in mean NO_3-N concentration in the control mesocosms at the start of the experiment contributed to a significant effect of time during the course of the experiment and the changes in mean NO_3-N concentration over time were significantly different between treatments (Table 6.3). No significant interaction effect of N × Si was found (Table 6.3). NO_3-N concentrations in Lake C2 were significantly higher than those in the control mesocosms at the end of the experiment (t >24.69, p = 0.002).

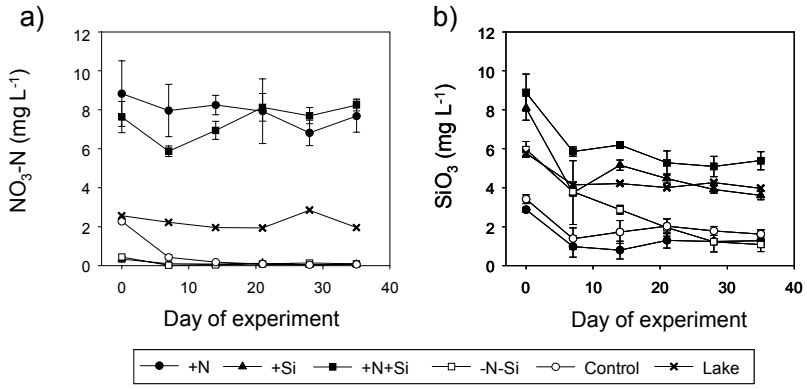

Figure 6.4 Mean concentrations of a) NO_3-N (mg L^{-1}); b) SiO_3 (mg L^{-1}) (±1 S.E.) in mesocosms and Lake C2 concentration during the experiment.

Throughout the experiment, the addition of Na_2SiO_3 resulted in significantly higher mean Si concentrations in +Si and +N+Si treatments (Figure 6.4, Table 6.3) showing that the manipulation of Si concentrations was also successful. Significant differences in mean Si concentration between treatments occurred over time (Table 6.3). The mean Si concentration fell in all mesocosms at the start of the experiment, and although the mean Si concentration in the -N-Si treatment continued to fall until day 28, no significant interactions between time and treatment or N and Si were found

(Table 6.3). The mean concentration of Si in the control mesocosms was significantly lower than in the lake at the end of the experiment (t = 16.51, p = 0.004).

Table 6.3 Statistical significance of the effects of treatments on the mean SiO_3 and NO_3-N concentrations measured by RM-ANOVA. Significant ($p \leq 0.05$) pairwise comparisons found using the post hoc Games–Howell test are shown. '↑' indicates that the mean of the second treatment of the pair was higher (↓, lower). Pairwise abbreviations are given in Table 6.1. 'df', degrees of freedom; 'MS', mean square; 'Tr', nutrient treatment (tested by one-way RM-ANOVA); 't', time; 'Tr × t', treatment × time interaction (tested by one-way RM-ANOVA); N × Si, nitrogen × silica treatment interaction (tested by two-way RM-ANOVA). '*' indicates that the *F*-ratio and degrees of freedom was adjusted using the Greenhouse-Geisser correction.

Variable		df	MS	*F*	*p*	Significant pairwise comparisons
NO_3-N (mg L^{-1})	Tr	4,10	15.83	2257.01	**<0.001**	(+N,+Si↓) (+N,-N-Si↓) (+N,Ctrl↓) (+Si,+N+Si↑) (+N+Si,-N-Si↓) (+N+Si, Ctrl↓)
	t	2.01, 20.05	0.13	18.57*	**<0.001**	
	Tr × t	8.02, 20.05	0.05	6.89*	**<0.001**	
	N × Si	1, 8	0.001	0.113	0.746	
Si (mg L^{-1})	Tr	4,10	70.53	61.89	**<0.001**	(+N,+Si↑) (+N,+N+Si↑) (+N,Ctrl↑) (+Si,+N+Si↓) (+N,+Si,-N-Si↓) (+N+Si,Ctrl↓)
	t	1.99,19.90	55.67	30.93*	**<0.001**	
	Tr × t	7.96, 19.90	3.76	2.09*	0.087	
	N × Si	1, 8	2.216	1.657	0.234	

Nutrients

TP concentrations in Lake C2 ranged between 368 and 533 µg L^{-1}, and the mean was 446 (±28) µg L^{-1} during the experiment (Figure 6.5a). The mean TP concentration in the treatments at the beginning of the experiment ranged between 238 (±12) µg L^{-1} in the -N-Si treatment and 376 (±50) µg L^{-1} in the control mesocosms, although these differences were marginally insignificant (one-way ANOVA, $F_{(4,10)}$ = 3.25, p = 0.06). TP concentrations in the mescosms increased significantly throughout the experiment, except in those with NO_3-N added (time and nutrients × time, p = <0.001, Table 6.4).

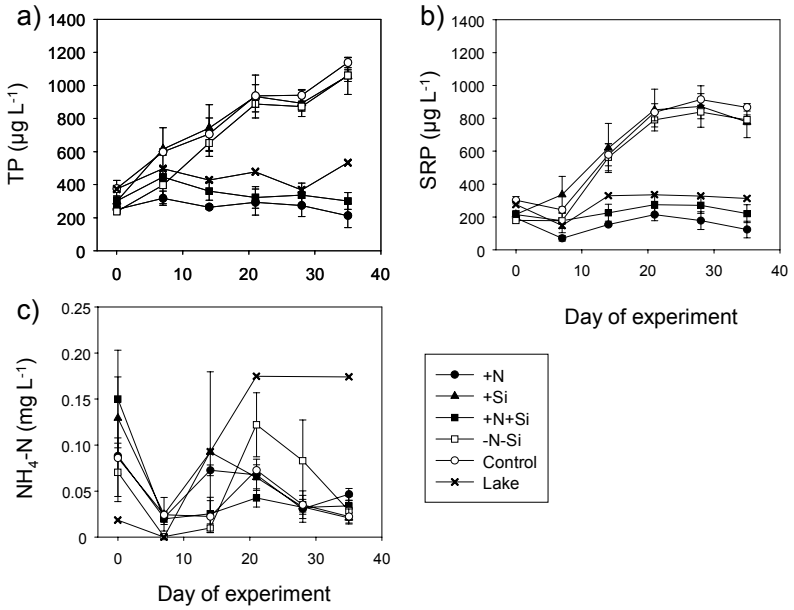

Figure 6.5 Mean concentrations (±1 S.E.) in mesocosms and concentration in Lake C2 during the experiment: a) TP (µg L^{-1}); b) SRP (µg L^{-1}); c) NH_4-N (mg L^{-1}).

Table 6.4 Statistical significance of treatments on the mean TP, SRP and NH_4-N concentrations measured by RM-ANOVA. Presentation as in Table 6.3.

Variable		df	MS	*F*	*p*	Significant pairwise comparisons
TP (µg L^{-1})	Tr	4, 10	1053472.74	17.25	**<0.001**	(+N, -N-Si↑) (+N, Ctrl↑) (+N+Si, -N-Si↑) (+N+Si, Ctrl↑) (+N+Si, -N-Si↑)
	t	5, 50	420210.79	72.36	**<0.001**	
	Tr × t	20, 50	85964.50	14.80	**<0.001**	
	N × Si	1, 8	63.96	0.001	0.976	
SRP (µg L^{-1})	Tr	4, 10	907120.60	15.72	**<0.001**	(+N,+Si↑) (+N,-N-Si↑) (+N, Ctrl↑) (+Si, +N+Si↓) (+N+Si, Ctrl↑) (+N+Si,-N-Si↑)
	t	2.21, 22.01	1170098.87	151.57*	**<0.001**	
	Tr × t	8.83, 22.01	154346.46	19.99*	**<0.001**	
	N × Si	1, 8	1696.34	0.26	0.877	
NH_4-N (µg L^{-1}) $log_{(x+1)}$	Tr	4,10	0.20	0.46	0.765	
	t	2.57, 25.72	3.59	11.02*	**<0.001**	
	Tr × t	10.29, 25.72	0.55	1.68*	0.138	
	N × Si	1, 8	0.286	0.727	0.419	

Post-hoc pairwise comparisons showed that differences in TP concentration between +N and +N+Si treatments and other mescosms were significant (Table 6.4). The N × Si interaction was insignificant (Table 6.4). At the end of the experiment, the TP concentration in the lake was not significantly different to that in the +N and +N+Si treatments, but was significantly lower than in the control treatments (t = 13.44, p < 0.006).

Changes in SRP concentration were similar to those observed for TP (Figure 6.5b). SRP concentrations in Lake C2 did not change substantially during the monitoring period. Significant differences in mean initial SRP concentrations existed between the control mesocosms and the +N and -N-Si treatments ($F_{(4,10)}$ = 7.093, p = 0.006). Over the duration of the experiment, the mean SRP concentrations were higher in the mesocosms without NO_3-N additions and the increases over time were significant (Table 6.4). In mesocosms with NO_3-N additions mean SRP concentrations did not increase and the time × treatment interaction was highly significant (p ≤0.001, Table 6.4). The interaction between N and Si was not significant (Table 6.4). SRP concentrations were significantly higher in the control mesocosms than in the lake at the end of the experiment (t = 17.35, p <0.005)

The concentration of NH_4-N in Lake C2 during the experiment ranged between 0 and 0.18 mg L^{-1} (Figure 6.5c) and the mean was 0.09 (±0.04) mg L^{-1}. NH_4-N concentrations in Lake C2 were slightly lower than those in the mesocosms at the beginning of the experiment although the difference was not significant. Mean NH_4-N concentrations did not vary significantly between mesocosms during the experiment (Table 6.4). There was a significant effect of time on mean NH_4-N concentrations although not of treatment × time (Table 6.4). In all treatments. NH_4-N concentrations decreased between 0 and 7 days and then rose until 21 days after the start of the experiment. Between 21 and 35 days, NH_4-N concentrations in all mesocosms decreased. NH_4-N concentrations in the lake were significantly higher than in the control mesocosm (t = 13.07, p <0.006) at the end of the experiment.

Major ions and alkalinity

The concentrations of Mg^{2+} were similar amongst treatments and between the lake and the mesocosms (Figure 6.6a). The mean Mg^{2+} concentration in Lake C2 was 1.5 (±0.1) meq L^{-1}, and ranged from 1.1 to 1.9 meq L^{-1}. Significant increases in Mg^{2+} concentrations occurred in all treatments and Lake C2 from the start of the experiment until day 21 (Table 6.5). The mean Mg^{2+} concentration was marginally significantly higher in the -N-Si treatment compared to the control mesocosms (p = 0.05, Table 6.5). Isolation did not significantly affect Mg^{2+} concentrations the control mesocosms compared to Lake C2.

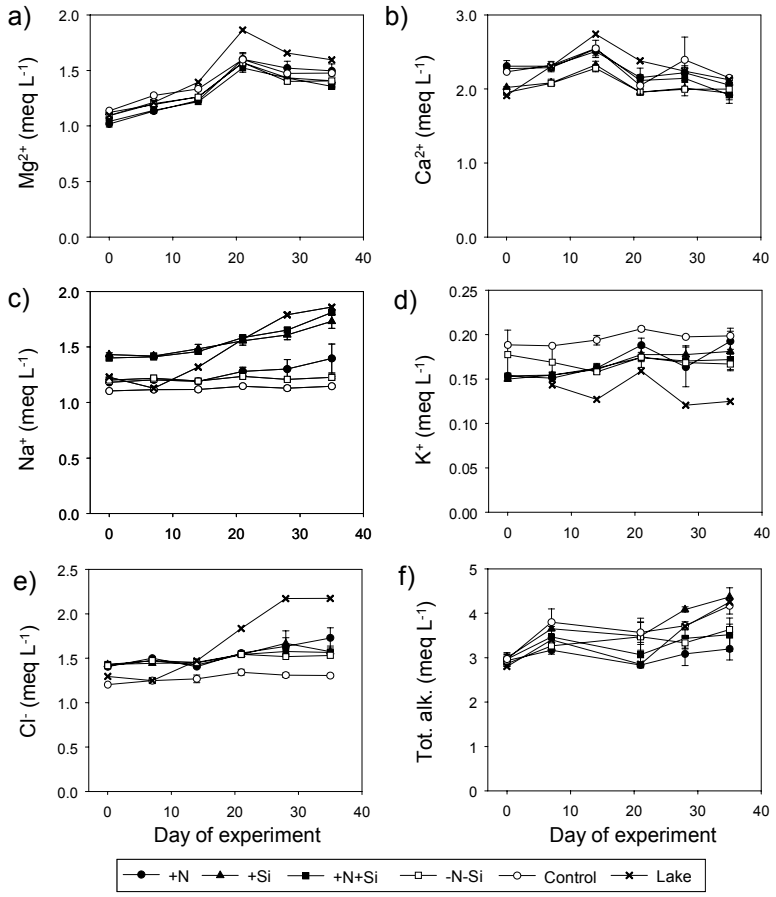

Figure 6.6 Mean concentrations of major ions and alkalinity (±1 S.E.) in mesocosms and Lake C2 concentration during the experiment: a) Mg^{2+} (meq L^{-1}); b) Ca^{2+} (meq L^{-1}); c) Na^{+} (meq L^{-1}), d) K^{+} (meq L^{-1}); e) Cl^{-} (meq L^{-1}); f) total alkalinity (meq L^{-1}).

Table 6.5 Statistical significance of the effects of treatments on the mean concentrations of major ions and mean total alkalinity, measured by RM-ANOVA. Presentation as in Table 6.3.

Variable		df	MS	*F*	*p*	Significant pairwise comparisons
Mg^{2+}	Tr	4,10	0.21	3.81	**0.039**	(Ctrl, -N-Si↓)
(meq L^{-1})	t	2.62, 26.23	0.97	219.22*	**<0.001**	
	Tr × t	10.49, 26.23	0.01	1.84*	0.101	
	N × Si	1, 8	0.02	2.22	0.175	
Ca^{2+}	Tr	4, 10	0.23	3.78	**0.040**	(+N, -N-Si↓)
(meq L^{-1})	T	1.87, 18.67	0.93	18.88*	**<0.001**	
	Tr × t	7.46, 18.67	0.39	0.80*	0.604	
	N × Si	1, 8	0.02	0.23	0.646	
Na^{+}	Tr	4, 10	0.69	62.00	**<0.001**	(Ctrl, +Si↑)
(meq L^{-1})						(Ctrl, +N+Si↑)
						(Ctrl, -N-Si↑)
	t	1.31, 13.12	0.34	34.69*	**<0.001**	
	Tr × t	5.25, 13.12	0.05	5.21*	**0.007**	
	N × Si	1, 8	0.004	0.308	0.594	
K^{+}	Tr	4, 10	0.003	5.03	**0.018**	
(meq L^{-1})	t	2.61, 10.42	0.002	5.86	**0.005**	
	Tr × t	10.42, 26.05	0.000	0.90	0.548	
	N × Si	1, 8	0.000	0.02	0.889	
Cl^{-}	Tr	4, 10	0.006	21.73	**<0.001**	(Ctrl, +N↑)
(meq L^{-1})						(Ctrl, +Si↑)
						(Ctrl, +N+Si↑)
						(Ctrl, -N-Si↑)
	t	1.33, 13.25	0.009	19.26*	**<0.001**	
	Tr × t	5.30, 13.25	0.001	1.33*	0.310	
	N × Si	1, 8	0.000	1.49	0.257	
Total	Tr	4, 10	0.002	4.29	**0.028**	
alkalinity	t	2.84, 28.44	0.019	17.61*	**<0.001**	
(meq L^{-1})	Tr × t	11.38, 28.44	0.002	1.42*	0.215	
	N × Si	1, 8	0.092	0.37	0.561	

The mean Ca^{2+} concentration in Lake C2 during the experiment was 2.3 (±0.1) meq L^{-1}, and ranged from 1.9 to 2.7 meq L^{-1} (Figure 6.6b). Ca^{2+} concentrations in all mesocosms increased slightly between the beginning of the experiment and day 14, and then fell during the rest of the experiment. Time had a significant effect on mean Ca^{2+} concentrations. The mean Ca^{2+} concentration was significantly higher in the +N than the -N-Si treatments (Table 6.5), probably because of the addition of $Ca(NO_3)_2$. No significant difference was found between control mesocosms and Lake C2 at the end of the experiment.

Na^{+} concentrations in Lake C2 gradually increased throughout the experiment, from 1.2 meq L^{-1} at the beginning of the experiment to 1.9 meq L^{-1} by day 35 (Figure 6.6c).

Na^+ concentrations increased significantly during the experiment (Table 6.5), although the greater increase in mesocosms with either Si or NO_3-N additions contributed towards a significant interaction between time and treatment (Table 6.5). Na^+ concentrations were probably increased by the addition of Na_2SiO_3 to the +Si and +N+Si mesocosms. Na^+ concentrations were significantly higher in Lake C2 than the control mesocosms at the end of the experiment (t = >88.75, p <0.0001).

K^+ concentrations were lower in Lake C2 than in the mesocosms during the experiment (Figure 6.6d), and ranged between 0.12 and 0.16 meq L^{-1} (mean, 0.13 ± 0.01 meq L^{-1}). In the mesocosms, mean K^+ concentrations increased significantly over time (Table 6.5). Mean K^+ concentrations were higher in the control treatments, although the difference was marginally significant (p = 0.08). K^+ concentrations in the control mesocosms were significantly higher than in Lake C2 (t = 9.16, p = 0.012).

Cl^- concentrations in Lake C2 were slightly lower than the mean concentrations observed in the mesocosms at the beginning of the experiment, and by the end of the experiment they were higher than the mean concentrations in all treatments except +N+Si (p ≤0.05, Figure 6.6e). A slight but significant rise in Cl^- concentrations occurred which was consistent across all mesocosms during the experiment. The nutrient × time interaction was insignificant (Table 6.5). The mean concentration of Cl^- was lower in the control mesocosms than all others throughout the duration of the experiment (Table 6.5), and at the end of the experiment was significantly higher in the lake than the control mesocosms (t = 33.64, p <0.001).

During the experiment total alkalinity increased significantly over time in all mesocosms (Table 6.5). Pairwise comparisons between treatments revealed only marginally higher total alkalinity in +N and +Si treatments (p = 0.068) and no significant treatment × time or N × Si interactions were found (Table 6.5). Total alkalinity did not differ significantly between the lake and the control mesocosms.

Physico-chemical variables

The water temperature in Lake C2 was slightly higher than in the mesocosms (Figure 6.7a). Lake C2 water temperature ranged between 17.0 and 19.9°C during the experiment (mean 18.2 (±0.5) °C). No significant difference existed in temperature

between the treatments during the experiment and the N × Si interaction was insignificant. Water temperatures varied significantly over time (Table 6.6).

At the start of the experiment, specific conductivity in Lake C2 was similar to that in the mesocosms (Figure 6.7b). After 7 days, specific conductivity in Lake C2 increased and remained higher than that in the mesocosms until the end of the experiment. Specific conductivity increased significantly during the experiment in all mesocosms (Table 6.6). Treatment had a significant effect on mean specific conductivity, and was generally highest in the +N+Si and +N treatments although no significant pair-wise differences were found. No significant Tr × t or N × Si interactions were found. At the end of the experiment, specific conductivity was significantly higher in the control treatments compared to the lake (t = 8.37, p = 0.014).

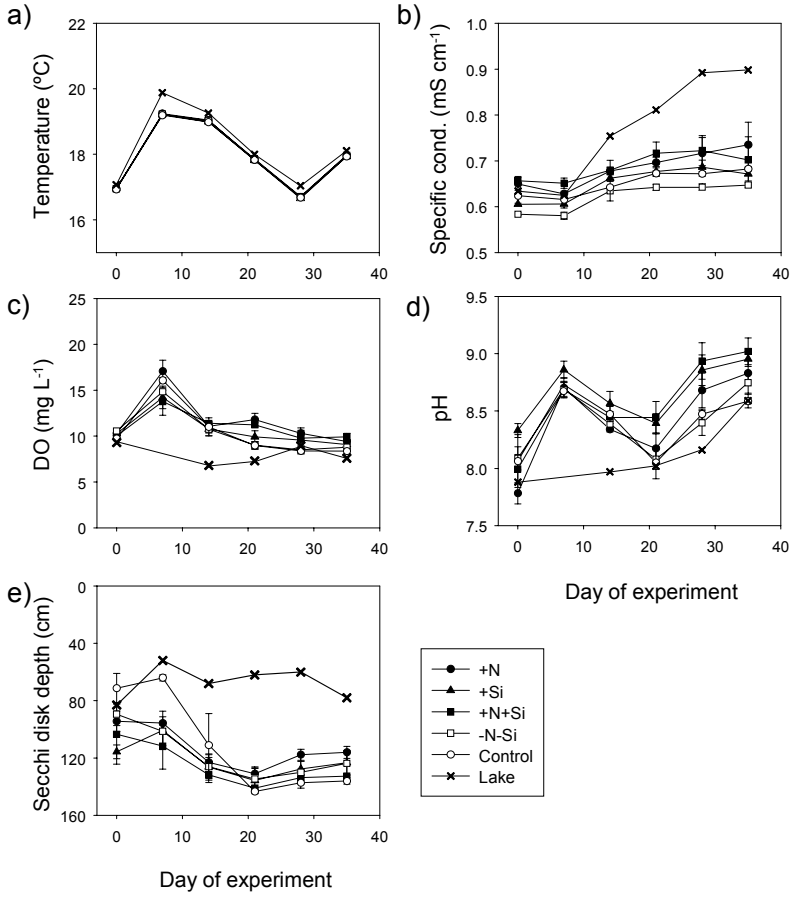

Figure 6.7 Mean physico-chemical parameters (±1 S.E.) in mesocosms and measurements from Lake C2 during the experiment: a, temperature (°C); b, specific conductivity (mS cm^{-1}); c, DO (mg L^{-1}); d, pH; e, Secchi disk depth (cm).

Table 6.6 Statistical significance of the effects of treatments on the mean concentrations of physico-chemical variables, measured by rm-ANOVA. 'Temp', temperature; 'Spec. cond.'; specific conductivity. Presentation as in Table 6.3

Variable		df	MS	*F*	*p*	Significant pairwise comparisons
Temp. (°C)	Tr	4, 10	0.004	0.44	0.779	
	t	5, 50	16.07	11886.13	**<0.001**	
	Tr × t	20, 50	0.001	0.41	0.984	
	N × Si	1, 8	0.003	0.387	0.551	
Spec. cond ($mS\ cm^{-1}$)	Tr	4, 10	0.13	6.13	**0.009**	
	t	1.81,18.06	0.43	18.20*	**<0.001**	
	Tr × t	7.23, 18.06	0.001	0.37*	0.912	
	N × Si	1, 8	0.003	1.18	0.309	
DO ($mg\ L^{-1}$)	Tr	4, 10	5.01	3.02	0.071	
	t	5, 50	75.54	73.94	**<0.001**	
	Tr × t	20, 50	1.83	1.79	0.098	
	N × Si	1, 8	3.47	1.77	0.220	
pH	Tr	4, 10	0.27	3.79	**0.040**	
	t	2.19, 29.94	3.20	56.56*	**<0.001**	
	Tr × t	8.78, 29.94	0.11	1.86*	0.115	
	N × Si	1, 8	0.04	0.52	0.491	
Secchi depth (cm)	Tr	4, 10	0.02	2.75	0.089	
	t	2.44, 24.38	0.17	22.08*	**<0.001**	
	Tr × t	9.75, 24.38	0.02	1.92*	0.092	
	N × Si	1, 8	338.00	0.88	0.377	

In Lake C2, DO concentrations were generally less than the mean DO concentrations observed in the mesocosms (Figure 6.7c). The range of DO was 6.8–9.3 $mg\ L^{-1}$ and the mean concentration was 7.8 $mg\ L^{-1}$. Mean DO concentrations in the mesocosms increased between the start of the experiment, then declined significantly (Table 6.6) to approximately 9.1 $mg\ L^{-1}$ at the end of the experiment. There was no significant interaction between N and Si. The mean DO concentration in the control mesocosms at the end of the experiment did not differ significantly from the lake.

The pH of Lake C2 ranged between 7.8 and 8.7 during the experiment (Figure 6.7d). Changes in pH over time were significant (Table 6.6): pH in all of the mesocosms rose at the start of the experiment before falling briefly and then increasing until the end of the experiment. Treatment had a marginally significant effect on mean pH although no significant pair-wise comparisons were found and the interaction between N and Si

was not significant (Table 6.6). At the end of the experiment, pH did not vary significantly between the lake and control mesocosms.

No significant effect of treatment was found on the mean Secchi depth (Table 6.6, Figure 6.7e). The mean Secchi disk depth at the beginning of the experiment was not significantly different between treatments. Secchi disk depths increased significantly over time (Table 6.6). After the start of the experiment, Secchi disk depths increased in all treatments. The Secchi disk was visible on the lake bed in most mesocosms after 21 days until the end of the experiment. No significant interaction between treatment and time or N and Si was found. The Secchi depth in Lake C2 was greatest at the start of the experiment, then decreased and remained at approximately 60 cm from day 7. At the end of the experiment, Secchi disk depth was significantly greater in the control treatment than in Lake C2 (t = 17.79, p = 0.003).

6.3.2 Phytoplankton

Chlorophyll-*a* concentrations at the beginning of the experiment (46 µg L^{-1}) were similar in Lake C2 compared to the mesocosms (Figure 6.8a). Mean chlorophyll-*a* concentrations at the start of the experiment (15-26 µg L^{-1} in treatments and 40 µg L^{-1} in the control mesocosms) did not vary significantly between treatments and control mesocosms. Mean chlorophyll-*a* concentrations varied significantly over time (Table 6.7). After the first sampling the mean chlorophyll-*a* concentration in all treatments increased sharply and decreased between day 7 and day 14, to less than 15 µg L^{-1} in all mesocosms. Mean concentrations of chlorophyll-*a* remained low for the rest of the experiment. RM-ANOVA showed that +Si treatments had significantly lower mean chlorophyll-*a* concentrations than the +N and -N-Si treatments (Table 6.7). No significant treatment × time or N × Si interaction was found. An increase in chlorophyll-*a* concentration occurred between day 0 and day 7 in Lake C2, and then chlorophyll-*a* concentrations began to decrease towards the end of the experiment. At the end of the experiment, chlorophyll-*a* concentrations in Lake C2 were significantly higher than in the control mesocosms (t = 10.51, p = 0.009).

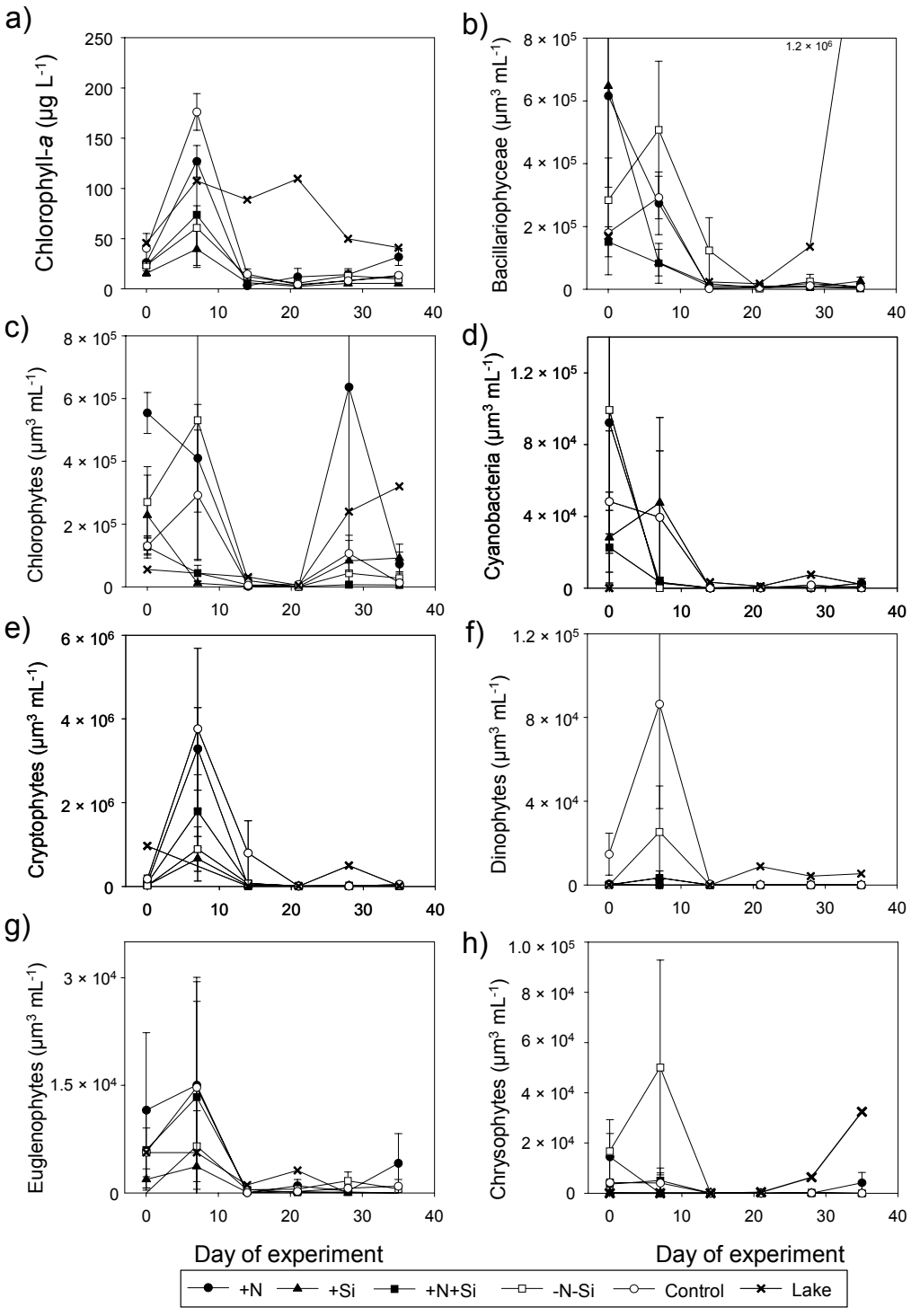

Figure 6.8 Mean abundance (µm^3 mL^{-1} ±1 S.E.) of total phytoplankton abundance and phytoplankton groups in mesocosms, and measurements from Lake C2 during the experiment: a) chlorophyll-*a* concentration (µg L^{-1}); b) Bacillariophyceae; c) chlorophytes; d) cyanobacteria; e) cryptophytes; f) dinophytes; g) euglenophytes; h) chrysophytes (all µm^3 mL^{-1}).

Table 6.7 Statistical significance of the effects of treatments on the mean biovolume of phytoplankton groups, measured by RM-ANOVA. Presentation as in Table 6.3

Variable		df	MS	*F*	*p*	Significant pairwise comparisons
Chlorophyll-a ($\mu g\ L^{-1}$)	Tr	4, 10	0.39	5.42	**0.014**	(+N, +Si↓), (-N-Si, +Si↓)
	t	5, 50	2.46	42.67	**<0.001**	
	Tr ×t	20, 50	0.14	1.51	0.121	
	N × Si	1, 8	0.03	0.50	0.500	
Bacillariophyceae ($\mu m^{-3}\ mL^{-1}$)	Tr	4, 10	0.46	1.09	0.413	
	t	2.2, 22.02	24.64	20.73*	**<0.001**	
	Tr ×t	8.81, 22.02	1.29	1.08*	0.413	
	N × Si	1, 8	1.01	3.82	0.086	
Chlorophytes ($\mu m^{-3}\ mL^{-1}$)	Tr	4, 10	151.69	2.63	0.098	
	t	2.23, 22.32	25.92	23.34*	**<0.001**	
	Tr ×t	8.93, 22.32	1.60	1.45*	0.229	
	N × Si	1, 8	0.35	0.64	0.447	
Cyanobacteria ($\mu m^{-3}\ mL^{-1}$)	Tr	4, 10	2.52	1.13	0.396	
	t	5, 50	11.84	5.14	**0.001**	
	Tr ×t	20, 50	1.97	0.85	0.640	
	N × Si	1, 8	0.04	0.02	0.907	
Cryptophytes ($\mu m^{-3}\ mL^{-1}$)	Tr	4, 10	1.06	2.08	0.159	
	t	5, 50	12.65	26.20	**<0.001**	
	Tr ×t	20, 50	0.39	0.81	0.695	
	N × Si	1, 8	0.35	0.97	0.353	
Euglenophytes ($\mu m^{-3}\ mL^{-1}$)	Tr	4, 10	0.72	0.27	0.892	
	t	5, 50	3.03	1.28	0.289	
	Tr ×t	20, 50	2.79	1.18	0.312	
	N × Si	1, 8	1.47	0.84	0.386	
Dinophytes ($\mu m^{-3}\ mL^{-1}$)	Tr	4, 10	40.08	60.69	**<0.001**	(+N, Ctrl↑) (+N+Si, Ctrl↑)
	t	2.45, 24.45	19.254	8.20*	**0.001**	
	Tr ×t	9.78, 24.45	5.23	2.23*	0.053	
	N × Si	1, 8	0.002	0.003	0.958	
Chrysophytes ($\mu m^{-3}\ mL^{-1}$)	Tr	4, 10	0.15	0.49	0.745	
	t	5, 50	17.98	12.46	**<0.001**	
	Tr ×t	20, 50	2.04	1.41	0.162	
	N × Si	1, 8	0.66	0.35	0.573	

The treatments did not have a significant effect on the mean total biovolume of diatoms (Figure 6.8b, Table 6.7). *S. acus* and small centric diatoms were the most abundant diatoms in the mesocosms. At the beginning of the experiment, no significant differences were found between the mean total biovolume of diatoms in each treatment. The mean biovolume of diatoms changed significantly over time, decreasing after the start of the experiment and remaining low until the end (Table 6.7). In Lake C2, *Synedra acus*, *Cocceneis* sp. and *Aulacoseira* spp. were the dominant taxa throughout the experiment. The abundance of all diatoms decreased

during the first 21 days of the experiment then increased, and although the biovolume of diatoms was higher in Lake C2 than the control mesocosms, the difference was only marginally significant at the end of the experiment (t = 3.89, p = 0.06).

No significant effect of nutrients was found on the mean biovolume of chlorophytes. The mean biovolume of chlorophytes changed significantly over time (Figure 6.8c, Table 6.7). Chlorophytes (mainly *Scenedesmus* spp. *Tetraedron* spp, *Tetrastrum* spp. and *Chlamydomonas* spp.) in the mesocosms generally decreased in abundance at the start of the experiment, and increased between days 21 and 28. Changes over time were not significantly different between treatments (no significant interaction). No significant difference was found between the biovolume of chlorophytes in the control mesocosms and Lake C2 at the end of the experiment.

No significant effect in mean cyanobacterial biovolume was observed between treatments (Figure 6.8d, Table 6.7). At the beginning of the experiment, no significant differences were found in the mean biovolume of cyanobacteria between treatments. The mean biovolume of cyanobacteria in the mesocosms varied significantly over time (Table 6.7). The biovolume of cyanobacteria decreased in all treatments, except +Si, between the first and second sampling. This was associated with a reduction in the biovolume of *Microcystis* sp. *Oscillatoria limnetica* became the most abundant cyanobacteria after 14 days, although the total biovolume of cyanobacteria remained substantially less abundant than other phytoplankton groups. There was no significant difference between cyanobacteria biovolume in Lake C2 and the control mesocosms at the end of the experiment.

The effect of treatment and the interaction between treatment and time had no significant effect on the mean biovolume of cryptophytes (Table 6.7, Figure 6.8e). Cryptophytes were abundant during the experiment and represented by *Cryptomonas* sp. and *Rhodomonas* sp., and were strongly correlated with chlorophyll-*a* concentration (r_s = 0.697, $p \leq 0.001$). The mean biovolume of cryptophytes was strongly affected by time (Table 6.7). The abundance of cryptophytes increased between the start of the experiment and day 7, then fell in all treatments, and remained low until the end of the experiment. At the end of the experiment cryptophyte biovolume did not differ significantly between Lake C2 and the control mesocosms.

The mean biovolume of dinophytes (Figure 6.8f) was significantly affected by the treatments (Table 6.7). Pair-wise differences revealed a higher mean biovolume of dinophytes in control mesocosms than +N and +N+Si treatments. At the beginning of the experiment, the mean biovolume of dinophytes was greater in the control mesocosms compared to treatment mesocosms ($F_{(4,10)}$ = 6.61, p = 0.007). The mean biovolume of dinophytes varied significantly over time. Dinophytes increased in the control and -N-Si treatments between the start of the experiment and day 7 then fell and remained low until the end of the experiment. At the end of the experiment, no dinophytes were found in the control mesocosms, and 5.4 × 10^3 µm^3 mL^{-1} in Lake C2.

No significant effect of nutrients was observed on the mean biovolume of euglenophytes and their biovolume did not vary significantly over time (Figure 6.8, Table 6.7). The only genus of euglenophyte that was observed in the mesocosms throughout the experiment was *Trachelomonas* sp. The biovolume of euglenophytes at the end of the experiment was not significantly different between the control mesocosm and Lake C2.

Treatment, and the treatment × nutrient interaction, did not significantly affect the mean biovolume of chrysophytes (Figure 6.8h, Table 6.7). Chrysophytes in Lake C2 were represented only by *Mallomonas* sp. The biovolume of *Mallomonas* sp. in Lake C2 increased during the experiment. The only genus of chrysophyte found during the experiment in the mesocosms was *Mallomonas* sp. The mean biovolume of *Mallomonas* sp. changed significantly over time (Table 6.7). Between the start of the experiment and day 7, the mean biovolume of *Mallomonas* sp. in the -N-Si treatments increased. During the same period, the mean biovolume of *Mallomonas* sp. decreased in the +N treatment and remained. *Mallomonas* sp. was rarely found in the mesocosms after day 7, and none were found in the control treatment at the end of the experiment.

Phytoplankton community structure

The first axis identified by PCA accounted for 30.4% of the variance in the phytoplankton dataset. The second axis accounted for an additional 7.2% of phytoplankton community variance, and axis 3 and four were associated with a total of 11.2% of variance. In total, axes 1 to 4 accounted for 48.8% of the phytoplankton community variability.

The plot of phytoplankton species (Figure 6.9a) shows all phytoplankton genera forming one cluster. No clear differences in sample scores between treatments were evident from the PCA plot. At the start of the experiment, phytoplankton of all mesocosms plotted towards the bottom right of the PCA plot (Figure 6.9b), with high axis 1 scores and low axis 2 scores (Figure 6.10a and b). This suggests a phytoplankton community associated with chlorophytes such as *Tetrastrum*, *Coelastrum* and *Pediastrum*, and the cyanobacterium *Microcystis* sp. Sample scores on axis 2 increased between day 0 and 7, reflecting the substantial increase in the biovolume of the cryptophytes *Cryptomonas* sp. and *Rhodomonas* sp. Higher axis 2 scores also suggest an increase in the importance of the chlorophyte genera *Monoraphidium*, *Tetraedron*, and *Scenedesmus*. Increases in axis 2 scores are also likely to be related to the decrease in abundance of *Microcystis*, *Scenedesmus*, *Coelastrum* and *Tetrastrum*. From day 14 until the end of the experiment, sample scores on both axes 1 and 2 strongly clustered towards the right of Figure 6.9b. This was probably a reflection of a decrease in the overall abundance of phytoplankton.

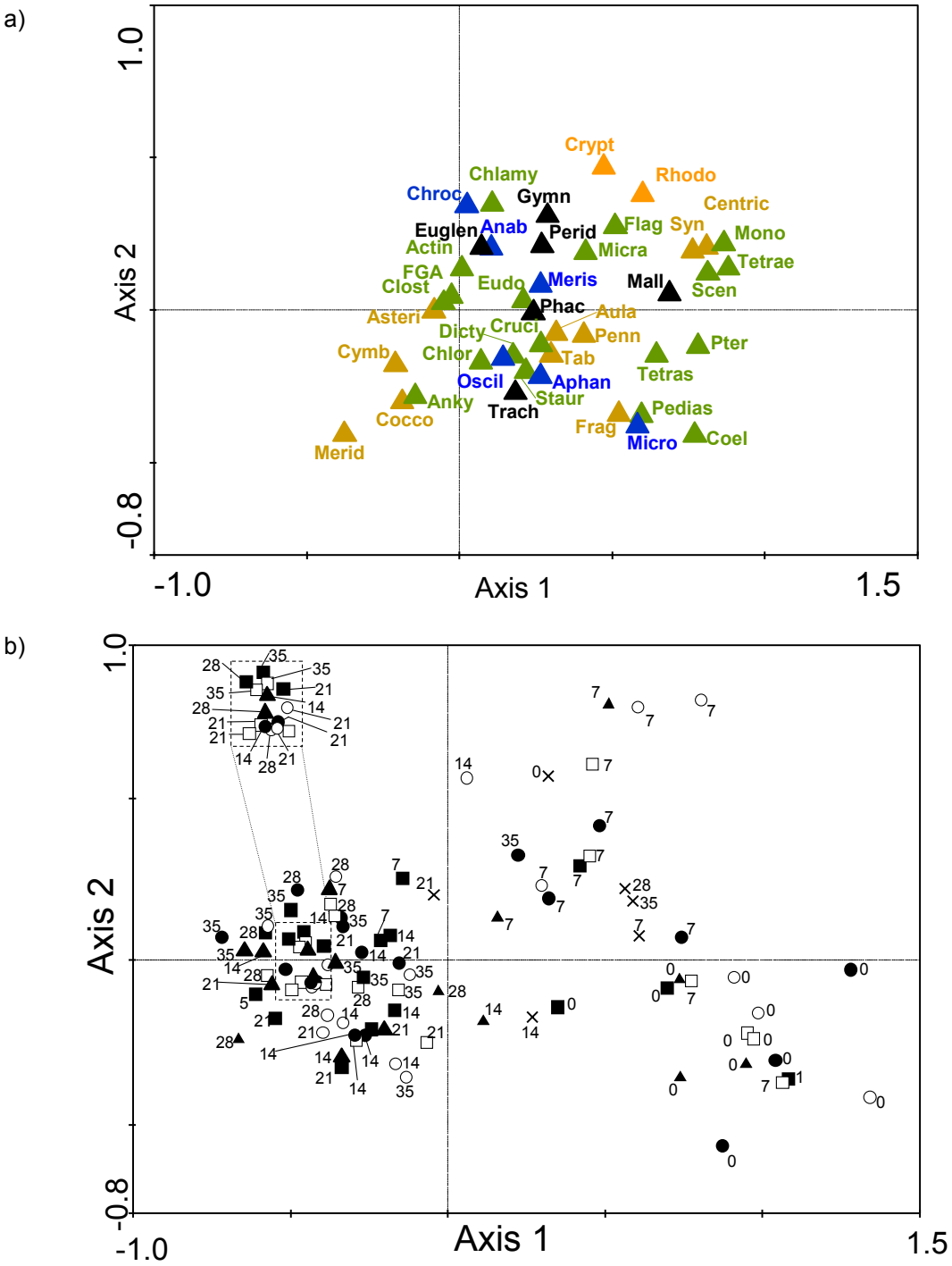

Figure 6.9 PCA of phytoplankton genera and samples, with scaling focused on inter-species correlation. a) Phytoplankton genera colour-coded by group: brown, Bacillariophyceae; green, chlorophytes; orange, cryptophytes; blue, cyanobacteria; black, others (including chrysophytes, euglenophytes and dinophytes). For abbreviations see Table 6.8. b) Samples coded by treatment (●, +N; ▲, +Si; ■, +N+Si; □, -N-Si; ○, Control and ×, Lake C2) and annotated with sample day

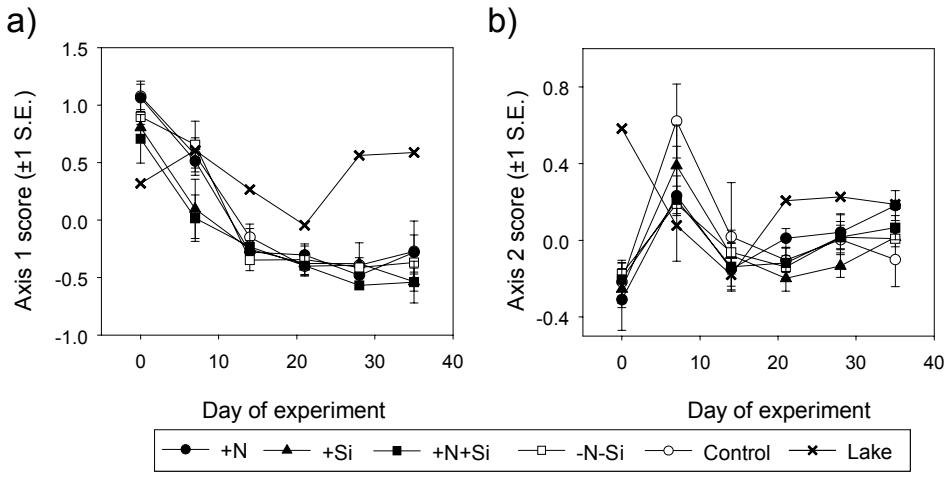

Figure 6.10 Mean phytoplankton PCA scores (±1 S.E.) during the mesocosm experiment for all treatments and Lake C2. a) axis 1; b) axis 2.

Table 6.8 Abbreviations used in Figure 6.9a.

Abbreviation	Genus
Bacillariophyceae	
Asteri	*Asterionella*
Aula	*Aulacoseira*
Centric	Centric species (*Stephanodiscus* / *Cyclotella type*)
Cocco	*Cocconeis*
Cymb	*Cymbella*
Frag	*Fragilaria*
Merid	*Meridion*
Penn	Pennate diatoms
Syn	*Synedra*
Tab	*Tabellaria*
Chlorophytes	
Actin	*Actinastrum*
Anky	*Ankyra*
Chlamy	*Chlamydomonas*
Chlor	*Chlorella*
Clost	*Closterium*
Coel	*Coelastrum*
Cruci	*Crucinigella*
Dicty	*Dictyosphaerium*
Eud	*Eudorina*
FGA	Filamentous Green Algae
Flag	Unidentified green flagellates
Micra	*Micratinium*
Mono	*Monoraphidium*
Pedias	*Pediastrum*
Pter	*Pteromonas*
Scen	*Scenedesmus*
Staur	*Staurastrum*
Tetrae	*Tetraedron*
Tetras	*Tetrastrum*

Table 6.8 contd.

Abbreviation	Genus
Cryptophytes	
Crypt	*Cryptomonas*
Rhodo	*Rhodomonas*
Cyanobacteria	
Anab	*Anabaena*
Aphan	*Aphanizomenon*
Chroc	*Chroococcus*
Meris	*Merismopedia*
Micro	*Microcystis*
Oscil	*Oscillatoria*
Others	
Euglen	*Euglena*
Gymn	*Gymnodinium*
Mall	*Mallomonas*
Perid	*Peridinium*
Phac	*Phacus*
Trach	*Trachelomonas*

6.3.3 Zooplankton

No significant differences in the mean total number of zooplankton (Figure 6.11a) occurred between treatments and the N × Si interaction was insignificant. There was no difference in total zooplankton biomass at the beginning of the experiment between treatments. A significant effect of time was observed (Figure 6.9). The total number of zooplankton in all mesocosms was low at the beginning of the experiment, increased to day 14 and then declined for the remainder of the experiment. Total zooplankton abundance in Lake C2 was lower than in the mesocosms during the experiment, and significantly lower than the control mesocosms at the end of the experiment (t = 4.69, p = 0.04).

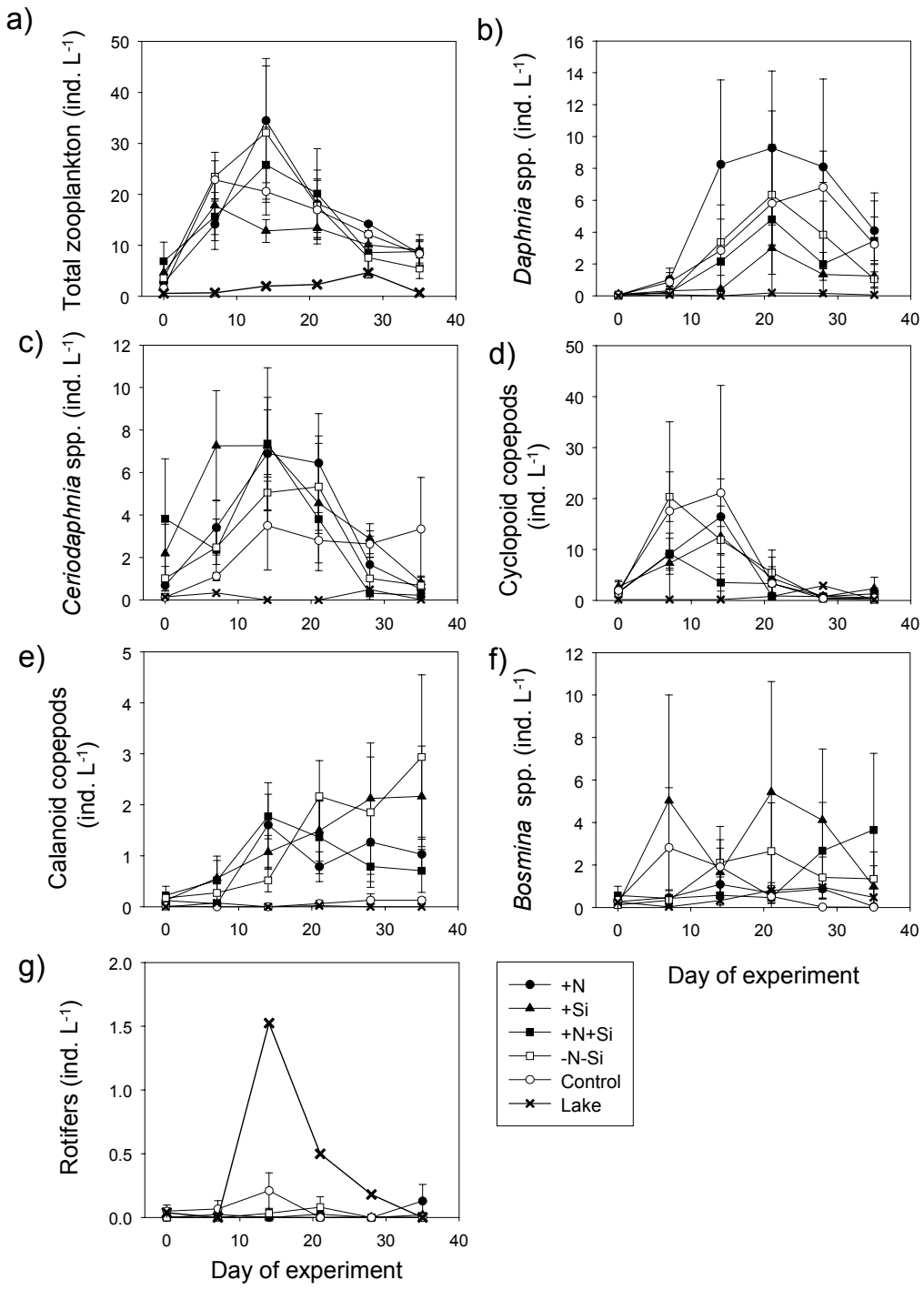

Figure 6.11 Mean abundance (ind. L^{-1}, ±1 S.E.) of total zooplankton abundance and zooplankton groups in mesocosms, and measurements from Lake C2 during the experiment: a) total zooplankton; b) *Daphnia* spp.; c) *Ceriodaphnia* spp.; d) cyclopoid copepods; e) calanoid copepods; f) *Bosmina* spp.; g) rotifers.

Table 6.9 Statistical significance of the effects of treatments on the mean abundance of zooplankton, measured by RM-ANOVA. Presentation as in Table 6.3.

Variable		df	MS	*F*	*p*	Significant pairwise comparisons
Total abundance (ind. L^{-1})	Tr	4, 10	45.38	0.299	0.872	
	t	2.33, 23.30	1992.29	14.65*	**<0.001**	
	Tr × t	9.32, 23.30	129.10	0.949*	0.505	
	N × Si	1, 8	0.000	0.000	0.992	
Daphnia spp. (ind. L^{-1})	Tr	4, 10	0.05	1.50	0.275	
	t	5, 50	1.03	17.22	**<0.001**	
	Tr × t	20, 50	0.05	0.85	0.642	
	N × Si	1, 8	0.03	0.45	0.520	
Ceriodaphnia sp. (ind L^{-1})	Tr	4, 10	9.60	1.40	0.304	
	t	3.01, 30.12	94.59	7.29*	**0.001**	
	Tr × t	12.05, 30.12	13.00	0.93*	0.536	
	N × Si	1, 8	0.12	1.24	0.298	
Cyclopoid. copepods (ind. L^{-1})	Tr	4, 10	0.07	0.38	0.821	
	t	5, 20	2.33	39.30	**<0.001**	
	Tr × t	20, 50	0.07	1.20	0.293	
	N × Si	1, 8	0.19	0.93	0.362	
Calanoid copepods (ind. L^{-1})	Tr	4, 10	0.22	3.91	**0.037**	(-N-Si, Ctrl↓)
	t	5, 20	0.18	7.07	**<0.001**	
	Tr × t	20, 50	0.03	1.10	0.375	
	N × Si	1, 8	0.000	0.000	0.996	
Bosmina spp. (ind. L^{-1})	Tr	4, 10	0.07	0.33	0.851	
	t	2.45, 24.49	0.25	1.63*	0.214	
	Tr × t	9.80, 24.49	0.15	0.95*	0.537	
	N × Si	1, 8	0.86	0.46	0.518	
Rotifers (ind. L^{-1})	Tr	4, 10	0.001	1.94	0.181	
	t	2.89, 28.87	0.001	0.78*	0.515	
	Tr × t	11.55, 28.87	0.002	1.22*	0.317	
	N × Si	1, 8	0.000	0.38	0.553	

No significant effect of treatment was observed on the mean abundance of *Daphnia* spp. and no significant interaction between N and Si was found (Table 6.9). The genus *Daphnia* was represented by the species *D. hyalina* and *D. pulex* in Lake C2 and mesocosms, although *D. hyalina* was more frequently found throughout the experiment. *Daphnia* spp. were generally rare in Lake C2, ranging in abundance between 0 and 0.2 ind. L^{-1} (Figure 6.11b). Significant changes over time were observed in the mean abundance of *Daphnia* spp. (Table 6.9). The mean abundance of *Daphnia* spp. increased from the beginning of the experiment in all treatments until day 21 and then began to decrease until the end of the experiment. Control mesocosms did not have a significantly different abundance of *Daphnia* spp. at the end of the experiment compared to the control mesocosms.

No significant effect of treatment or significant interaction between N and Si was detected on the mean abundance of *Ceriodaphnia* spp. (Table 6.9). *Ceriodaphnia* spp. varied significantly in abundance over the course of the experiment. Increases in abundance occurred at the start of the experiment in all treatments. From day 14 until the end of the experiment the mean abundance of *Ceriodaphnia* spp. decreased in all treatments (Table 6.9). *Ceriodaphnia* spp. were not frequently found in Lake C2 during the experiment (Figure 6.11c) and were not significantly more abundant in the lake than in the control mesocosms at the end of the experiment. The only other cladoceran found during the experiment was *Scapheloberis mucronata* which was most abundant between days 28 and 35.

No significant differences in mean cyclopoid copepod abundance were observed between treatments and the interaction between N and Si was insignificant (Table 6.9). Cyclopoid copepod abundance varied significantly over time (Table 6.9). In all treatments, the mean number of cyclopoid copepods increased from the beginning of the experiment until days 7 to 14 and then decreased for the remainder of the experiment. Cyclopoid copepods were found in Lake C2 throughout the experiment (Figure 6.11d). No significant difference was found between the mean abundance in control mesocosms and Lake C2.

Treatment had a significant effect on the mean abundance of calanoid copepods (Table 6.9); mean abundances were significantly higher in -N-Si mesocosms than the control mesocosms. No significant interaction occurred between N and Si. The mean abundance changed significantly over time (Table 6.9, Figure 6.11e). In +Si and -N-Si treatments, the mean abundance of calanoid copepods increased throughout the experiment. Calanoid copepods were virtually absent in the control mesocosms and Lake C2 during the experiment.

Bosmina longirostris was the most abundant member of the genus in the mesocosms. *B. coregoni* was also recorded in the mesocosms during the second week of the experiment. No significant effect of nutrients or interaction between N and Si was observed on the abundance of *Bosmina* spp. (Table 6.9). The mean abundance of *Bosmina* spp. did not vary significantly over time (Figure 6.11, Table 6.9) and was not significantly different in the mesocosms than in Lake C2. The abundance of *Bosmina*

spp. did not differ significantly between Lake C2 and the control mesocosms at the end of the experiment.

The mean abundance of rotifers was low in all treatments (Figure 6.11g). The rotifers found in zooplankton samples were either *Keratella* spp. or *Asplancha* spp. The mean abundance of rotifers did not vary significantly over time or between treatments, and no significant interaction between N and Si was found (Table 6.9). In Lake C2, *Asplancha* sp. was generally more abundant during the experiment. However, at the end of the experiment, rotifers were not recorded from either Lake C2 or the control mesocosms.

Zooplankton community structure

Axis 1 of the zooplankton PCA was associated with 38.2% of variance in the dataset, and the second axis was associated with an additional 24.3%. In total, the first four axes indentified by PCA were related to for 85.0% of variation in the zooplankton dataset.

Figure 6.12a shows a plot of the zooplankton taxa PCA scores. Daphnidae were split into two groups along axis 1, with higher axis 1 and two scores being associated with *Ceriodaphnia* sp and *D. hyalina*. Low axis 1 scores were associated with *S. mucronata* and *D. pulex*. The two groups of copepods observed during the experiment were also distinctly separated by the PCA. High axis 1 and low axis 2 scores were associated with cyclopoid copepods, and lower axis 1 and higher axis 2 scores corresponding to calanoid copepods. Chydoridae, rotifers and *Bosmina* spp. were all found near the centre of the PCA diagram.

A plot of sample scores (Figure 6.12b) shows no clear differences between the treatments. Samples from the beginning of the experiment had low axis 1 and two scores (Figure 6.13a and b), possibly related to the low abundance of all zooplankton groups during the first sample. Samples from the second and third weeks had higher axis 1 and two scores, driven by an increase in the abundance of cyclopoid copepods in all mesocosms. Increases in the axis 2 scores after 14 days reflect the increase in abundance of *D. hyalina* and *Ceriodaphnia* sp. Then decreases in the axis 1 scores

were observed, most likely a consequence of increases in the abundance of calanoid copepods. No significant effect of nutrients × time was found.

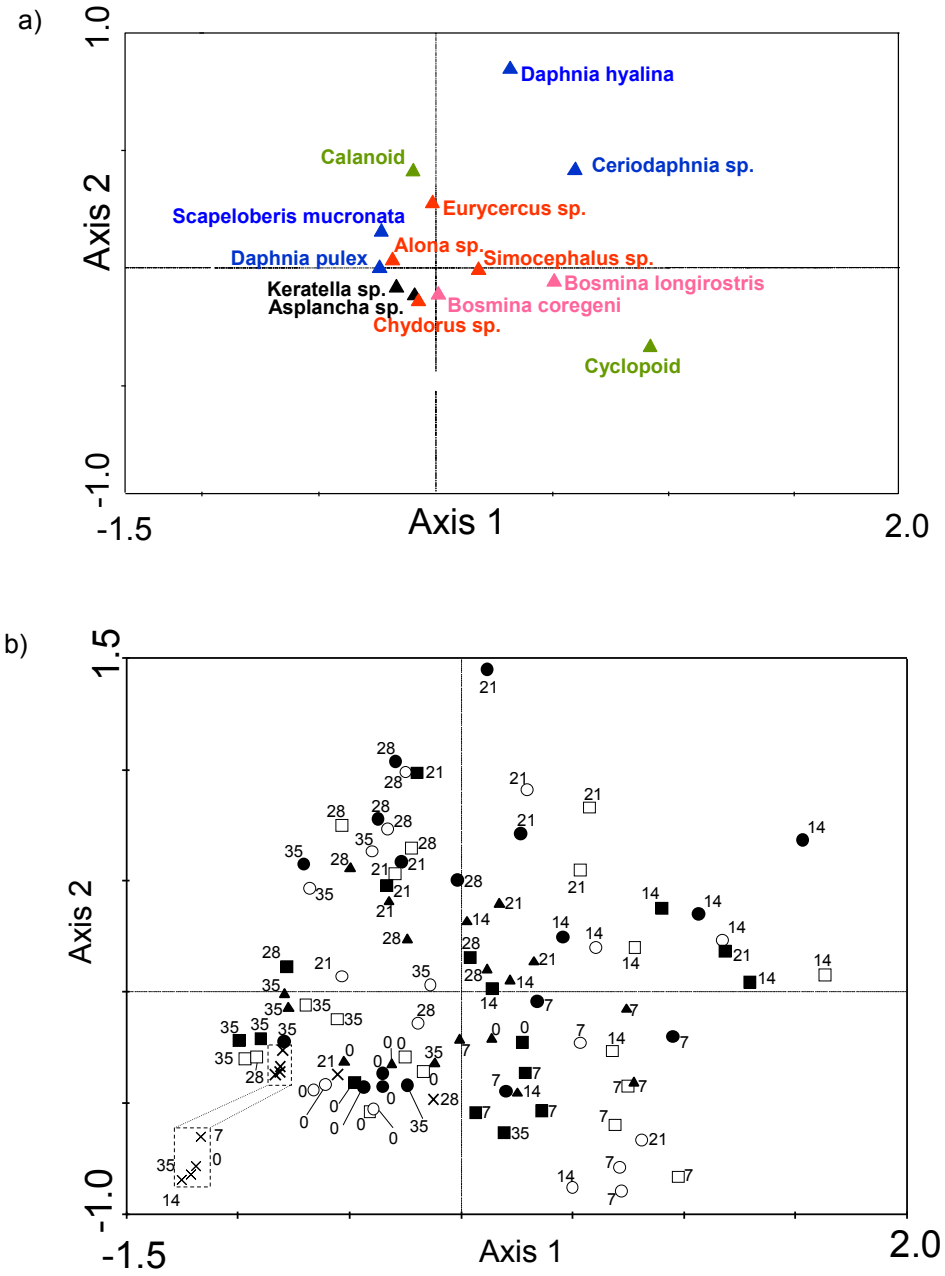

Figure 6.12 PCA of phytoplankton genera and samples, with scaling focused on inter-species correlation. a) Zooplankton genera are colour-coded by group: blue, Daphnidae; green, copepods; magenta, Bosminidae; black, rotifers and red, others. b) Samples coded by treatment (●, +N; ▲, +Si; ■, +N+Si; □, -N-Si; ○, Control and ×, Lake C2) and annotated with sample day.

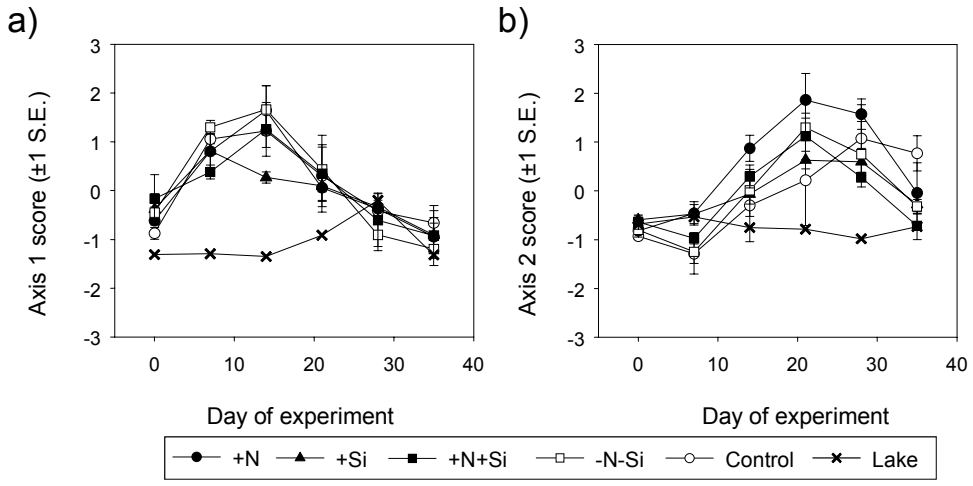

Figure 6.13 Mean zooplankton PCA scores (±1 S.E.) during the mesocosm experiment for all treatments and Lake C2. a) axis 1; b) axis 2.

6.4 Discussion

In mesocosms without N added and the control mesocosms, the concentration of TP and SRP increased during the experiment. The release of P from the sediments is likely to be responsible for the observed increases in P concentration (see discussion in section 3.9, page 187). P release from the sediments of eutrophic lakes is frequently observed (section 1.4.1, page 36; Søndergaard et al., 1999, 2001), although in-situ evidence of P release is scarce. During the mesocosm experiment the most likely cause of P release is probably related to anoxic conditions at the sediment surface which allows P to diffuse out of the sediment (Figure 1.8 page 39). The clear water and low phytoplankton biomass in the mesocosms suggests P sedimentation was high, possibly related to grazing of phytoplankton by zooplankton and sedimentation of zooplankton faecal pellets (e.g. Ferrante and Parker, 1977). The installation of the mesocosms involved some disturbance of sediments, which may have exaggerated the effects of P release. Resuspension of sediments has been found to be related to increases in P concentrations in shallow lakes (Zickler et al., 1956; Scheffer, 1998; Søndergaard et al., 2003).

A consequence of the manipulation of NO_3-N concentrations was the suppression of P release in the +N and +N+Si mesocosms. NO_3-N has been found to be associated

with reductions in P release in a variety of laboratory and field studies (see section 1.4.1, page 41, Andersen, 1982; Foy, 1986; Jensen and Andersen, 1992; Hansen et al., 2003). It is likely that the addition of NO_3-N to the +N and +N+Si treatments resulted in an increase of the available oxidised Fe which could be related to the suppression of P release during the experiment in these treatments. NO_3-N strongly oxidises lake sediments, because it is highly soluble in water and therefore is effective at supplying oxygen to the sediments (Hansen et al., 2003).

No significant effect of N addition on mean chlorophyll-*a* concentration or the mean biovolume of phytoplankton groups was observed. This contrasts with the increasing evidence that suggests N may limit total phytoplankton productivity (see section 1.4.2, page 42; Moss et al., 1994; Weithoff and Walz, 1999; Maberly et al., 2002; Camacho et al., 2003; James et al., 2003). This may particularly be the case in eutrophic lakes where denitrification rates can be high (Wetzel, 1983; Petzoldt and Uhlmann, 2006). Some mesocosms have shown that elevated concentrations of N may favour the growth of cryptophytes (e.g. Vrede et al., 1999; González Sagrario et al., 2005). NO_3-N additions to other mesocosms has been associated with rapid chlorophyll-*a* rises driven by cryptophytes (Présing et al., 1997). N enrichment has also been associated with a reduction in the biomass of cyanobacteria (e.g. Schindler, 1977) which may be favourable for lake management.

Since P was abundant in all N and Si manipulations it is surprising that no effect of NO_3-N was found. This may be associated with the NH_4-N concentrations in the mesocosms. NH_4-N is the preferred source of N for uptake by phytoplankton (e.g. Dortch, 1990) as assimilation of NO_3-N requires reduction to NH_4-N and is therefore less energy efficient (Wetzel, 1983). The high abundance of zooplankton in all mesocosm treatments suggests that recycling of NH_4-N could occur (Wen and Peters, 1994; Attayde and Hansson, 1999; Vanni, 2002), which may have been important for the growth of phytoplankton in the mesocosms. No significant differences were observed in NH_4-N concentrations between treatments which suggests that all mesocosm had similar NH_4-N concentrations. The negative correlation between NH_4-N and chlorophyll-*a* concentrations across all mesocosms (r_s = -0.226, p = 0.032) also suggests the uptake of NH_4-N by phytoplankton. Decreases in NH_4-N concentrations

were particularly evident between day 0 and day 7, when mean chlorophyll-*a* concentrations also increased sharply.

The supply of NH_4-N may explain the apparently anomalous finding that the biovolume of cyanobacteria decreased during the experiment in the mesocosms. It could be expected that conditions in the mesocosms would favour cyanobacteria growth, for example by low N:P ratios in the mesocosms without N addition (Shapiro, 1973; Smith, 1983). Abundant NH_4-N has been related to low cyanobacteria biovolume in other studies (e.g. Présing et al., 1997). Although zooplankton were abundant in the mesocosms, grazing is unlikely to have been related to the low cyanobacterial biovolume, as grazing zooplankton would probably consume smaller species in preference to cyanobacteria (Agrawal, 1998; Reynolds, 2006). *Microcystis*, the most abundant cyanobacterium in the mesocosms, is resistant to grazing on account of its size and toxicity (Agrawal, 1998). The ability of cyanobacteria to regulate their buoyancy (Reynolds et al., 1987) might avoid the effects of sedimentation experienced by other phytoplankton groups.

The addition of Si to the mesocosms caused a decrease in chlorophyll-*a* concentrations. Mean chlorophyll-*a* concentrations in the +Si treatment were lower than the +N and -N-Si treatments. This difference was probably associated with the comparatively smaller increase in chlorophyll-*a* concentration observed between the first and second sampling in the +Si mesocosms. The mean chlorophyll-*a* concentration between day 0 and 7 increased by 24 μg L^{-1} in the +Si treatment and 101 and 37 μg L^{-1} in +N and -N-Si treatments respectively. This may have been related to the higher mean abundance of *Ceriodaphnia* sp. and *Bosmina* spp. in the +Si mesocosms, which could have reduced phytoplankton biomass by grazing. The small *F* ratio associated with this finding suggests that size of this effect was not large (Field, 2000; Gravetter and Wallnau, 2000). This is also supported by the similarity in mean chlorophyll-*a* concentrations between the +Si, and +N and -N-Si treatments, from the third sampling until the end of the experiment (Figure 6.8a).

The addition of Si did not significantly increase the biomass of diatoms in contrast to the expectation that higher Si concentrations may favour diatom growth. Some ambiguity exists as to the relationship between concentrations of Si measured in water

and the abundance of diatoms. For example, Bailey-Watts (1976) found no clear relationship between planktonic diatom abundance and Si concentrations, and suggested that this may be related to the sinking of diatom cells (see section 1.4.3, page 48). It is possible that the biovolume of diatoms in the mesocosms was adversely affected by sedimentation. Sinking is a loss process which can significantly affect phytoplankton that are unable to regulate their buoyancy (Reynolds et al., 1982; Reynolds, 2006). Diatoms in particular are susceptible to sinking, as sinking rates have been estimated to be at least twice as fast as non-siliceous phytoplankters of similar geometric shape (Sommer, 1991), and this may have contributed towards the decline in their biomass. Sedimentation may have been favoured by the increased height of the mesocosm walls because of the increase in water depth in Lake C2 (Figure 6.3c and d) which would have reduced the potential for wind mixing of the water column.

The only significant change in phytoplankton community was the higher dinophyte biovolume in the control mesocosms compared to the +N and +N+Si. This finding mainly reflects the increase in dinophyte biomass in the control mesocosms at the beginning of the experiment. An increase in dinophyte biovolume may be associated with the increase in P concentration in the control treatment (>220 µg L^{-1} between day 0 and 7). Romo and Villena (2005) found that dinophytes were more abundant in mesocosms enriched with both N and P but Gonzalez-Sagrario et al. (2005) found no response of dinophytes to N and P enrichment. *Peridinium* sp. may be associated with the phytoplankton of small nutrient-enriched ponds (Reynolds et al., 2002) which suggests that their growth may have been favoured by higher P concentrations. However, dinophytes were the rarest phytoplankton group during the experiment, and even a statistically significant increase in their biovolume appeared to have little effect on the overall phytoplankton community composition (Figure 6.10).

The abundance of zooplankton did not vary significantly between treatments. The only significant difference found between treatments was a marginally higher abundance of calanoid copepods in the -N-Si than the control treatment. This is probably because calanoid copepods were transferred in the water from Lake I2, where they were considerably more abundant, particularly during summer 2007 when the experiment took place (>10 ind. L^{-1} in Lake I1, Figure 3.40c; <0.5 ind. L^{-1} in Lake C2 Figure 3.16c).

The mean abundance of calanoid copepods for the entire experiment was higher in all mesocosms that were filled with water from Lake I2 than the control mesocosms, although not sufficiently greater to be significantly different.

Grazing by zooplankton is likely to have had a substantial effect on the phytoplankton in the mesocosms. Since the enclosures probably protected the zooplankton from zooplanktivorous fish, populations of zooplankton were able to increase substantially compared to the abundances observed in Lake C2. The findings of Chapter 4 suggested that grazing possibly exerted a strong influence on abundance of phytoplankton in Lake C2, and across all mesocosms, negative correlations were observed between chlorophyll-*a* concentrations and *Daphnia* spp. (r_s = -0.470, $p \leq 0.001$). Grazing has often been suggested as a potentially important modifying factor between nutrient concentrations and total phytoplankton biomass (e.g. Carvalho, 1994; Moss et al., 1994; Lau and Lane, 2002; Carvalho and Kirika, 2003). The potential for a strong grazing effect in this experiment may have been high because of the dominance of small and easily grazed species which are considered to be susceptible to grazing by zooplankton, such as cryptophytes (Talling, 2003), small centric diatoms and the chlorophytes *Ankyra* and *Chlamydomonas* (Reynolds, 2006). This finding demonstrates the importance of fish in Lake C2 for regulating the trophic cascade.

A particularly important finding of this experiment for the management of Attenborough Nature Reserve is that there is a large potential for the release of P from the sediments. This confirms the findings of Chapter 3 (see section 3.9, page 187) and the Vollenweider modelling discussed in section 5.5.2 (page 256). The significantly higher P concentrations in the mesocosms without NO_3-N added than the lake suggests that isolation of the lakes, and removal of the supply of NO_3-N from the River Erewash (section 5.4.1, page 246; section 5.5.1, page 254) may favour high in-lake P concentrations. Nutrient reduction measures in shallow lakes may include the removal of sediments in order to reduce the pool of P that may be released to the water column (e.g. Jeppesen et al., 1991; Annadotter et al., 1999; Søndergaard et al., 2000), and this technique might reduce the magnitude of P release from the sediments, if only in the short-term (Phillips et al., 1999). The diversion of the River Erewash and subsequent reduction in NO_3-N loading to the connected lakes may also increase P

release by reducing oxidation of the lake sediments. Adding NO_3-N into connected lakes may therefore appear to be an effective method for reducing P release after the diversion of the River Erewash. However, significant costs are involved with the addition of NO_3-N to lakes (Foy, 1986). Furthermore, the addition of NO_3-N to lakes is also unwise given the evidence that diverse and stable vegetation communities require very low NO_3-N concentrations (1–2 mg L^{-1}), and that high N concentrations may ultimately delay recovery from eutrophication (González Sagrario et al., 2005; James et al., 2005).

The absence of fish in the mesocosms was probably the main cause of the large population of zooplankton during the experiment. Juvenile fish are often zooplanktivorous (such as perch, *Perca fluviatilis*, which becomes piscivorous), whilst planktivorous fish (including roach, *Rutilus rutilus*, and bream, *Abramis brama*) are likely to prey on zooplankton throughout their life (Brönmark and Hansson, 1998). Several studies have suggested that predation by planktivorous fish is important for determining the size and structure of the zooplankton community (Brooks and Dodson, 1965; Lammens, 1999; Attayde and Hansson, 2001). Meijer et al. (1990) found that the exclusion of a mixture of cyprinid fish (bream, carp and small roach) from sections of a shallow lake resulted in a higher abundance of large-bodied zooplankton (e.g. *Daphnia hyalina* and *D. magna*), although little difference in total zooplankton biomass was found. Hansson et al. (2007) identified the importance of young-of-the-year fish in ending the abundance of zooplankton during spring and therefore terminating the spring clear-water phase in a Swedish lake. In the connected lakes of Attenborough Nature Reserve, it is probable that a large community of planktivorous fish existed, therefore the isolation of the water column released zooplankton from top-down control by fish. Excluding fish from the mesocosms is likely to have affected the interpretation of the effects of nutrients and nutrient interactions on the phytoplankton community and is probably the largest limitation to the applicability of the experimental results for shallow lake management. Mesocosm experiments that have included fish as a treatment in the design have found that they alter phytoplankton and zooplankton biomass and community structure. Data gathered from a number of mesocosm studies show an increase in total phytoplankton biomass with the addition of planktivorous fish (e.g. Lynch and Shapiro, 1981; Proulux et al., 1996; Moss et al., 2004) although evidence for group level changes can be variable

between experiments (Lynch and Shapiro, 1981; Van de Bund et al., 2004). The addition of fish would have considerably increased the logistical challenge of replacing the water in the mesocosms after their construction. Sourcing fish may particularly challenging, and the death of fish in mesocosms (e.g. Romo and Villena, 2005) may complicate the treatments unless continually replaced as required. Since this was the first mesocosm experiment run in the Attenborough Nature Reserve, the experiment aimed to reduce the number of controlling variables and focus on bottom-up processes. However, future studies could consider using a fish treatment in addition to N and Si manipulations when simulating lake restoration. Experiments including fish may be more representative of natural lake ecosystems undergoing restoration and could provide more insight into the response of phytoplankton to N, Si and simulated restoration by reducing top-down control.

Some criticisms of the use of mesocosms in investigating ecological communities have been made (Carpenter and Kitchell, 1988; Schindler, 1998) although mesocosms may be complementary to long term monitoring and whole-lake experiments (Carpenter, 1997; Drenner and Mazumder, 1999). The argument that small-scale ecological experiments exclude some components of ecosystems (Carpenter, 1996) was evident during this experiment, for example through exclusion of fish which is a clear limitation to this experiment. Other components and processes may change so rapidly as to be unrealistic (Carpenter, 1996). For example, the rate of change of total phytoplankton biomass in the mesocosms appeared to reach approximately 22 μg L^{-1} day^{-1}, compared to an estimate (assuming a linear change over time) of up to 9 μg L^{-1} day^{-1} in the open lake. The small spatial and temporal scale of mesocosm experiments can give inconsistent results when compared to whole-lake scales (e.g. Carpenter and Kitchell, 1988). In this experiment, the small size of the mesocosms probably prevented a realistic stimulation of water column turbulence, which may have been significant for the potential of sedimentation of phytoplankton. Even if the results of mesocosm experiments are largely replicable (Schindler, 1998), extrapolating findings without appreciating the effects of spatial and temporal scales on mesocosm results may give erroneous management recommendations.

6.5 Conclusions

The results of the mesocosm experiment suggest that the isolating effect of the mesocosms was associated with changes in chemistry and in biology. P concentrations at the start of the experiment were not significantly less than the ambient lake concentration, and continued to increase throughout the experiment. Increases in P concentration may reflect the importance of flushing to prevent the accumulation of P in the overlying water column. This finding is supported by the findings of Chapter 5, which suggested that a reduction in water retention time was associated with decreased in-lake P concentration.

The rapid response of P concentrations in the mesocosms suggests that the diversion of the River Erewash and effective isolation of currently connected lakes will probably be associated with increased in-lake P concentrations. The addition of NO_3-N in the +N treatments suppressed P release, as has been observed in other field and experimental studies (e.g. Foy, 1986; Jensen and Andersen, 1992).

The isolation of lake water in the mesocosms was also associated with a rise in the abundance of zooplankton. The effect was observed across all mesocosms, and was probably a result of the exclusion of zooplanktivorous fish from the mesocosms. The increase in zooplankton is likely to be related to the sharp decline in total phytoplankton abundance after day 7 of the experiment. The response of phytoplankton to zooplankton grazing may have been greater than the effect of nutrient additions, which could explain why the effects of nutrients were related to few changes in chlorophyll-*a* concentration or mean phytoplankton group biovolumes. The decrease in chlorophyll-*a* concentrations due to grazing may have been particularly rapid due to the abundance of cryptophytes and chlorophytes which are susceptible to grazing by zooplankton (Reynolds, 2006).

Other mechanisms may have accounted for the decrease in abundance of other phytoplankton groups. Diatoms may have been adversely affected by sedimentation (Sommer, 1984) in addition to grazing by zooplankton as it is likely that there was little turbulence in the mesocosms. Diatom cells are particularly susceptible to sinking as they are unable to regulate their buoyancy and the siliceous cell walls increases their

density compared to other phytoplankton groups (Sommer, 1984). The abundance of cyanobacteria in the mesocosms also decreased. This may be related to the abundance of NH_4-N which could imply that N-fixing cyanobacteria would have no competitive advantage. As NH_4-N concentrations did not differ between treatments, this may explain why phytoplankton populations did not respond to the addition of NO_3-N to the mesocosms.

The findings of this chapter suggest that the isolation of currently-connected lakes from the River Erewash diversion will be associated with an increase in summertime P concentrations, consistent with the conclusions of Chapters 3 and 5 (see sections 3.9 and 5.9). Isolation of the lakes will prevent the flushing of P from the connected lakes and may result in P accumulating in the water column during the summer. Wind mixing may have been reduced by increasing the height of the mesocosm walls in response to the increase in water depth. This could have over-estimated sedimentation. Reducing the supply of NO_3-N to the connected lakes may reduce the oxidation of the sediments and increase P release further. However, it is unlikely that NO_3-N addition is a suitable technique for mitigating against internal P loading.

Chapter 7 Discussion and conclusions

This chapter examines the key findings that relate to each of the original aims of the thesis. The aims were:

1. to compare and contrast the chemical and biological structure and functioning of the shallow lakes at Attenborough Nature Reserve isolated from and connected to the River Erewash (discussed from page 330);
2. to investigate how flooding events may affect the biology and chemistry of Attenborough Nature Reserve (page 342);
3. to experimentally simulate lake restoration by river diversion and explore the effects of nitrogen and silica on ecosystem community structure at Attenborough Nature Reserve (page 346);
4. to explore the likely effects of diverting nutrient-rich water from shallow lakes, and evaluate these effects in the context of the Attenborough Nature Reserve (page 351).

The major findings are discussed in relation to the management of Attenborough Nature Reserve and more generally in the context of managing shallow eutrophic lakes. The chapter also discusses restoration techniques in addition to the diversion of the River Erewash which may benefit Attenborough Nature Reserve.

7.1 Compare and contrast the chemical and biological structure and functioning of shallow lakes isolated from and connected to nutrient-rich river inflows

7.1.1 Summary of major findings

- The River Erewash is an important external source of P, N and Si to the lakes and in-lake processing of nutrients appears to have varied among connected and isolated lakes;
- Isolated lakes I1 and I2 exist in the clear-water state and have abundant submerged macrophytes, but the connected lakes and lake I3 are turbid, devoid of macrophytes and dominated by phytoplankton;
- Bottom-up processes (nutrients) and flushing rate probably interact to strengthen top-down (grazing) control of phytoplankton during spring in the connected lakes compared to the isolated lakes;
- Internal release of P from the sediments probably occurrs in all lakes;
- The potentially beneficial effects of short WRT caused by connectivity to the River Erewash should be considered in management plans.

7.1.2 Nutrient supply and internal cycling

The River Erewash inflow is an important source of P for the connected lakes, as is often the case for shallow lakes which are connected to rivers (see section 5.4.1, page 246; e.g. Anderson et al., 1990; Beklioglu et al., 1999; Bootsma et al., 1999; Gulati and van Donk, 2002). Reducing the nutrient loading from such rivers is usually the initial action taken to restore lakes (Moss et al., 1996a; Gulati and van Donk, 2002). The diversion of the River Erewash is relatively unusual because the nutrient loading will be reduced by the removal of most of the inflow rather than by a reduction of point-source P discharges. Whether or not such a reduction in external loading has any significant effect on in-lake P concentrations is usually dependant on the degree of internal loading. This can be of particular importance in lakes with low WRTs. This is

because historically short WRTs may be associated with higher inflow of P-rich water and therefore higher P retention in the sediments (Jeppesen et al., 1991). P release represents a significant challenge for the management of Attenborough Nature Reserve. During the 35 days of the mesocosm experiment, TP concentrations in the mesocosms without NO_3-N added increased by over 800 µg L^{-1}, as discussed in section 6.4 (page 320). This suggests that a substantial pool of P exists that could be released. Experiences elsewhere suggest that summertime TP concentrations may not decline for around 15 years after a reduction in external P loading (Jeppesen et al., 2005b; Phillips et al., 2005; Søndergaard et al., 2005).

The River Erewash is an important source of NO_3-N because internal processing of NO_3-N (e.g. uptake by and sedimentation of phytoplankton and denitrification) reduces water-column concentration. Isolation of lakes from external N supplies is therefore associated with a reduction in NO_3-N concentration (e.g. Moss et al., 1986). The decline in NO_3-N concentration from Lake C1 to Lake C3 suggests increasing retention along the Erewash-connected lake chain (see Figure 3.2c and Figure 3.18c; section 3.9, page 187). NH_4-N is probably rapidly recycled within the connected lakes, by processes such as uptake by phytoplankton, regeneration by zooplankton, decomposition of organic matter and release from the sediments (Vanni and Temte, 1990; van Luijn et al., 1999; Présing et al., 2008). The importance of internal NH_4-N cycling is likely to be greater in lakes that are isolated from nutrient-rich rivers (e.g. Moss et al., 1996b). Using river diversion for lake restoration is significantly advantageous compared to catchment-scale nutrient control, as the diffuse nature of N loading means that N management is considerably more difficult than P loading management (see section 1.4.2, page 42 and section 7.4.3, 353; Petzoldt and Uhlmann, 2006). If internal P loading maintains an abundant P supply after river diversion, there may an increased risk of cyanobacterial blooms unless internal cycling of N is sufficient to prevent N limitation (e.g. Moss et al., 1986; 1996b). Evidence from lakes I1 and I2 suggests that N reaches limiting concentrations during the summer and cyanobacteria are proportionally more abundant in the phytoplankton community, although the total phytoplankton biomass was lower. For example, the relative abundance of cyanobacteria during summer 2006 in connected lakes was less than 25%, although in lakes I1 and I2 cyanobacteria, including heterocystous species such as *Aphanizomenon* sp. and *Anabaena* sp. comprised >50% of total phytoplankton

biovolume. A further discussion of cyanobacteria in Attenborough Nature Reserve is given in section 7.4.3 (page 353).

The supply of Si to the connected lakes was probably important for the in-lake concentration of Si, but internal release of Si may also have contributed to water-column Si. Inflowing concentrations of Si were significantly higher than those in all of the connected lakes during the monitoring period (see section 5.4.1, page 246). Diatoms were abundant in the phytoplankton during the spring and particularly during the late summer, when SiO_3 concentrations were depleted sufficiently to be potentially limiting for diatom growth. The biovolume of diatoms and SiO_3 concentrations were negatively correlated in the connected lakes (r_s = -0.565, p <0.001), reflecting the importance of diatoms for regulating water-column SiO_3 concentrations. Rapid increases in SiO_3 concentrations occurred (5 mg L^{-1} in 30 days were not uncommon, e.g. Figure 3.10, page 107) after the collapse of diatom populations. The presence of diatom frustules in the sediments (Sayer and Roberts, 2001) suggests that some Si is retained in the connected lakes. The nutrient budget developed in Chapter 5 (page 255) also showed that Si was retained during 2005 and 2006 in the connected lakes. Internal Si loading has been thought to exceed external loading in several studies (e.g. Bailey-Watts, 1976; Parker et al., 1977; Miretzky and Cirelli, 2004). In isolated lakes I1 and I2, Si concentrations were lower than in the connected lakes (section 3.9, page 187), suggesting that Si concentrations have declined since the isolation of those lakes from the River Erewash in 1981. Si concentrations in lakes I1 and I2 were not significantly correlated with the abundance of planktonic diatoms (r_s = -0.069, p = 0.557). It may be that the improvement in benthic light climate has improved the growth of benthic diatoms in lakes I1 and I2 consequently reduced the strength of correlation between water-column Si concentrations and planktonic diatom populations. Diversion of the River Erewash from the currently connected lakes may result in a reduction of planktonic diatoms after several years. For example, Philips et al. (2005) found that planktonic diatoms may have been replaced with a greater benthic diatom community 20 years after a reduction in external nutrient loading. However internal recycling of Si may be sufficient to prevent declines in planktonic diatom biomass in the short-term after the River Erewash diversion.

7.1.3 Alternative Stable States

Both elements of of the two states of the alternative stable states model (see section 1.3, page 29 Scheffer et al., 1993; Scheffer and Jeppesen, 1998) appeared to be evident in Attenborough Nature Reserve (Figure 7.1). Sayer and Roberts (2001) suggested that connectivity to the River Erewash may have induced a switch from clear to turbid water in the connected lakes. The results presented in this thesis show that the connected lakes are highly turbid (see section 3.9, page 187). Secchi disk depths during the monitoring period frequently reached less than 50 cm during the summer and the Secchi disk was never visible on the bottom of the connected lakes

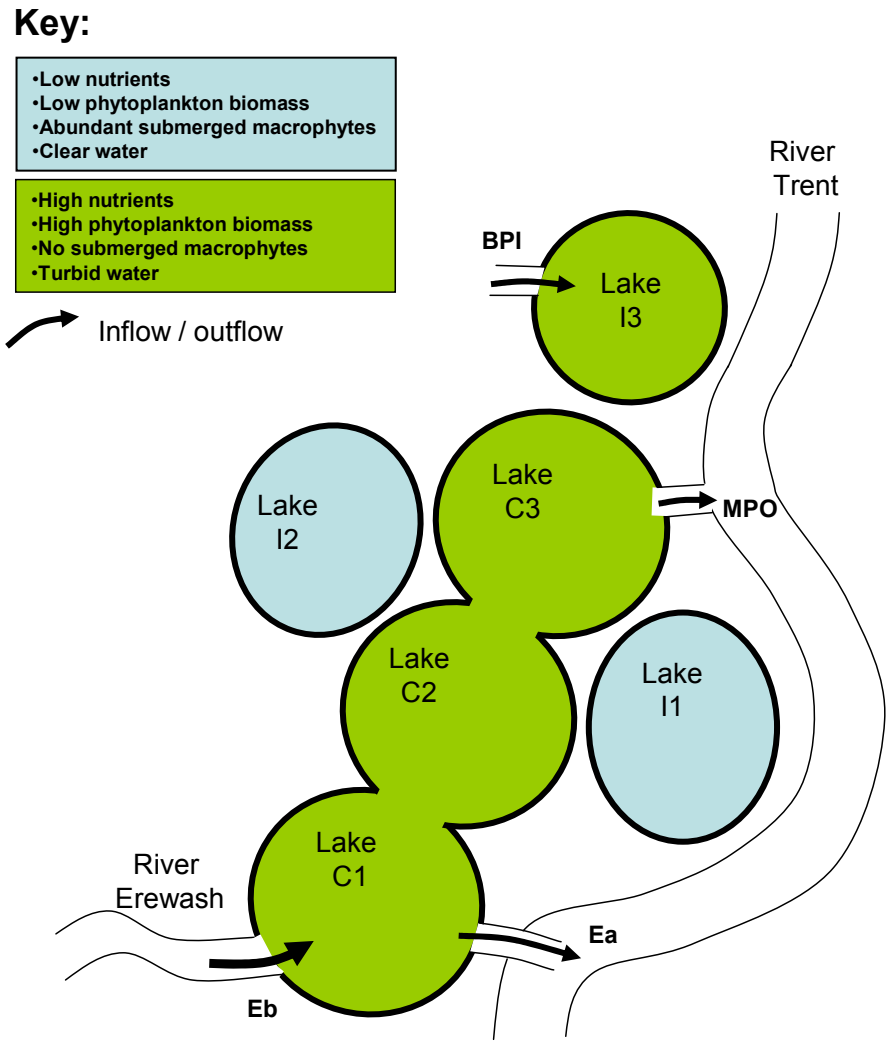

Figure 7.1 Schematic map of water clarity, phytoplankton biomass and submerged macrophytes in Attenborough Nature Reserve.

(see Figure 3.2l, Figure 3.10l, Figure 3.18l). During the summer of 2007, Secchi disk depths were similar to the previous years although chlorophyll-*a* concentrations were lower, which suggests that there was a greater influence of minerogenic turbidity associated with the flooding events. Secchi disk depth was more closely associated with the TSS concentration than with chlorophyll-*a* in the lakes where submerged vegetation was absent (connected lakes and Lake I3), than in the vegetation-dominated lakes I1 and I2 (Figure 7.2). This suggests that minerogenic turbidity may be important in maintaining the turbid state due by creating a poor underwater light climate, although the exact cause is difficult to interpret because TSS also includes the algal component of water turbidity. The estimates of Middelboe and Markager (1997) suggest that vegetation in the connected lakes would not grow at depths greater than 0.5 m because of the water turbidity. Therefore, except in some littoral areas, submerged vegetation is probably unable to grow in the connected lakes and Lake I3 because the water depth exceeds 0.5 m.

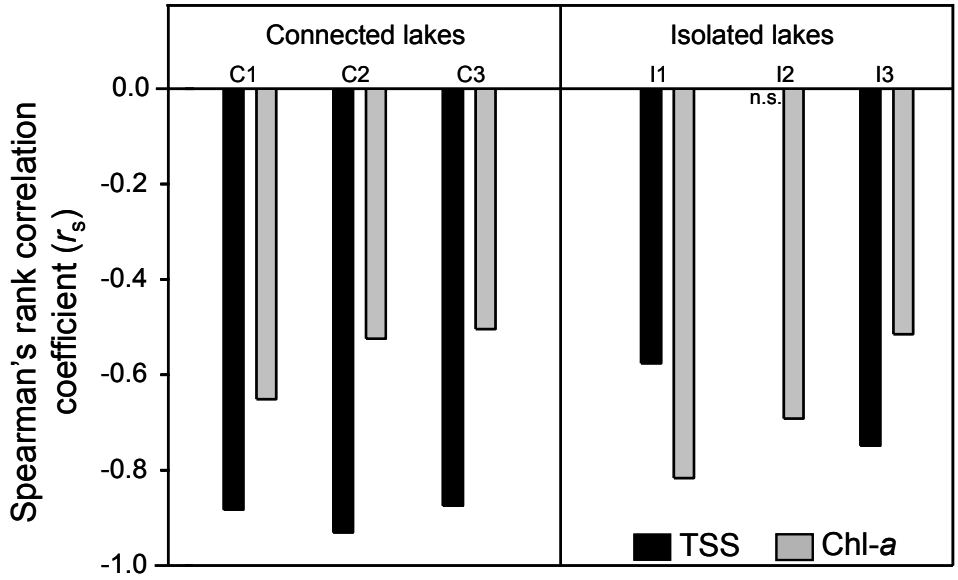

Figure 7.2 Spearman's rank correlation coefficients (r_s) between Secchi disk depth and chlorophyll-*a* (chl-*a*) and total suspended solids (TSS) concentrations during the monitoring period. All correlations shown are significant at $p \leq 0.01$. n.s., not significant ($p \geq 0.05$).

High algal growth associated with nutrient supply derived from the River Erewash may have reduced the resilience of the currently connected lakes, causing a switch to the turbid state when it was diverted in the currently connected lakes in 1972 (see section 1.3, page 29; Sayer and Roberts, 2001). The increase in the nutrient loading alone might have been sufficient to cause a decline in submerged vegetation, but stochastic events may have forced a shift between the clear and turbid states. Stochastic events are an underlying assumption of the alternative stable states model (Scheffer et al., 1993). Connectivity to the River Erewash may increase the likelihood of stochastic events occurring. For example, flood events may cause a large increase in minerogenic turbidity, such as was observed during January 2008, in the connected lakes which may contribute towards a collapse in vegetation. Water depth increases, which could be caused by flooding events, may also force a switch towards phytoplankton dominance (e.g. Blindow et al., 1993).

The turbid condition of Lake I3 is probably associated with the supply of nutrients from the inflow (BPI) (section 5.4.2, page 252). Nutrient concentrations in Lake I3 were generally higher than in the other isolated lakes but lower than those in the lakes connected to the River Erewash (section 3.9, page 187). Throughout the monitoring period, nutrient concentrations at BPI were significantly higher than the in-lake concentration suggesting that the inflow is a significant source of nutrients. It is possible that the original diversion of the River Erewash into the lakes may have caused a loss of submerged vegetation in Lake I3, although the lake was probably isolated from nutrient-rich water most of the time (Sayer and Roberts, 2001). The 'delta sanctuary' between lakes C3 and I3 (see Figure 2.1, page 74) may have helped protected Lake I3 from the River Erewash inflow.

In contrast, lakes I1 and I2 exist in a clear-water state. Secchi disk depths in both lakes were high, and the water turbidity was more strongly correlated with chlorophyll-*a* than TSS concentrations suggesting lower minerogenic turbidity than in the River Erewash connected lakes and Lake I3. Submerged vegetation was abundant during the summer months in both lakes. However, P concentrations in Lake I1 are at the higher end of P concentrations that may be expected for a clear-water lake. The maximum summer TP concentration during 2005 was approximately 400 µg L^{-1}, and in 2006 exceeded 600 µg L^{-1}. High TP concentrations suggest that the lake may be

susceptible to a switch to the turbid state (Moss et al., 1996a; Scheffer and Jeppesen, 1998) but winter NO_3-N concentrations of <0.6 mg L^{-1} during 2005 and 2006 are probably low enough to maintain stable and diverse vegetation communities (González Sagrario et al., 2005; James et al., 2005). Because of the high P concentrations, increases in NO_3-N concentration associated with very high flood events may jeopardise the stability of the clear-water state and cause a switch to turbid water. For example flooding events during the winter of 2007–2008 increased NO_3-N concentrations to over 1.0 mg L^{-1}, possibly related to the overtopping of water from Lake C2 (Figure 5.18, page 245). P concentrations in Lake I2 were lower during the monitoring period (maximum summer TP <200 µg L^{-1}) than in Lake I1 (see Figure 3.26a, page 134; and Figure 3.34a, page 147) which suggests that greater forcing would be required to switch the lake to a turbid state (Figure 7.3). NO_3-N concentrations were similarly low in both Lake I1 and Lake I2, and did not exceed 1 mg L^{-1} during the monitoring period (Figure 3.26c and Figure 3.34c). Positive feedback mechanisms associated with the presence of submerged macrophytes are likely to be important for the maintenance of clear-water in Lake I1 and I2 because of the potential for high phytoplankton growth. Zooplankton grazing appeared to limit phytoplankton biomass (Table 4.15, page 219) which may have been maintained by the refuge effect of submerged macrophytes (e.g. Timms and Moss, 1984; Scheffer, 1998). An abundance of submerged macrophytes may have shaded phytoplankton (Van Donk and Van de Bund, 2002) and may have contributed to the sedimentation of phytoplankton (Barko and James, 1998). *Chara* spp., which were found in both lakes I1 and I2, may release allelopathic substances (e.g. Mulderij et al., 2006; 2007) and further hinder phytoplankton growth.

Lakes I1 and I2 may be useful references for nutrient-rich and artificial shallow lakes. This is important as defining appropriate reference conditions for artificial lakes in order to meet the requirements of the Water Framework Directive (European Community, 2000) is difficult. The high error associated with modelling type-specific lake reference types suggests that a site-specific approach is more robust (e.g. Carvalho et al., 2009), further supporting the use of lakes I1 and I2 for establishing reference conditions for the Attenborough Nature Reserve. High P concentrations and low NO_3-N concentrations are potential outcomes of diverting the River Erewash, which suggests that the current states of lakes I1 and I2 (abundant submerged

macrophyte community and clear water) may be an appropriate target for the restoration of the connected lakes. In support of this, palaeoecological evidence suggests that submerged macrophytes were abundant in Lake C2 before the River Erewash was diverted into the currently connected lakes (Sayer and Roberts, 2001).

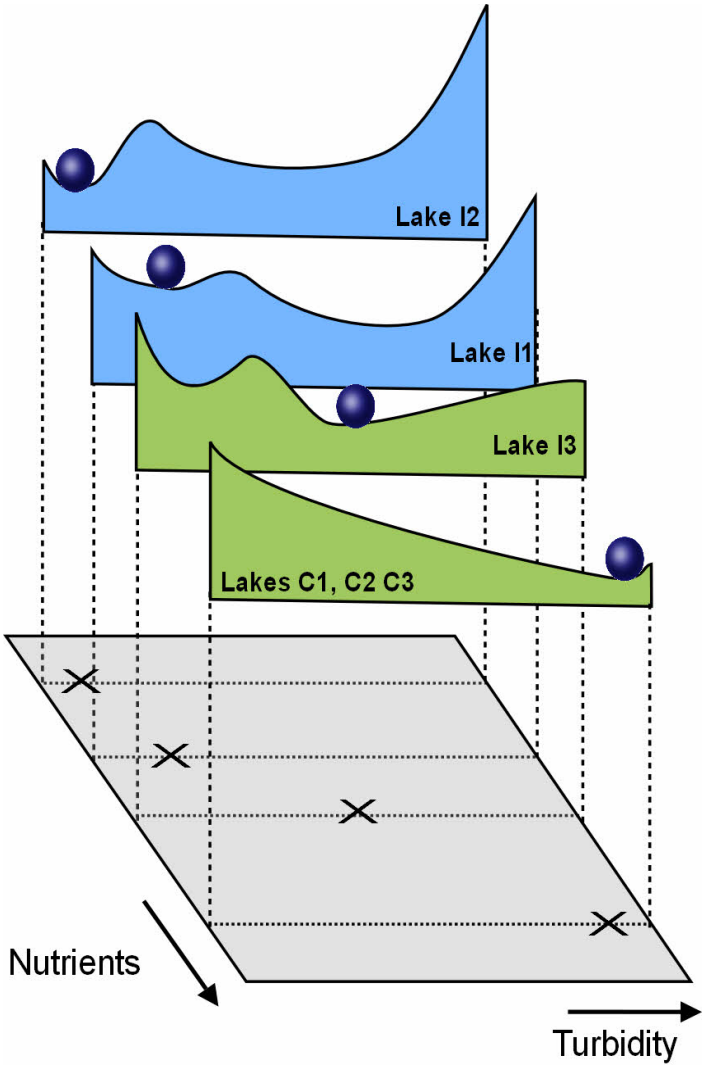

Figure 7.3 Stability landscapes schematically representing nutrient concentration and turbidity in the lakes of Attenborough Nature Reserve. The balls represent the current state of the lakes, and the depth of the valley in which it rests represents the hypothesised stability of the current state. Colour-coding is derived from Figure 7.1. Based on Scheffer (1998).

7.1.4 Top-down and bottom-up control of phytoplankton

A combination of top-down and bottom-up control of phytoplankton occurred in the lakes of Attenborough Nature Reserve during the monitoring period, as discussed in Chapter 4 and summarised in Table 4.15, page 219. During spring, zooplankton grazing was probably related to the clear-water period in the connected lakes. Chlorophyll-*a* concentrations were reduced to less than 10 μg L^{-1} and Secchi disk depths increased to around 150 cm. In contrast, a spring clear-water period was not clearly evident in isolated lakes I1 and I2, although zooplankton were significantly related to phytoplankton community composition. In Lake I3, a spring clear-water period did not occur and zooplankton were not identified as significant in determining either phytoplankton biomass or community structure (Chapter 4).

The well-defined clear-water period observed in the connected lakes might be explained by the dominance of smaller, easily-grazed phytoplankton typical of lakes with short WRTs and high nutrient loading (Figure 7.4). The physical effects of short WRT may have favoured the development of small phytoplankton species that are able to reproduce sufficiently quickly to replace their losses from outwash (Brook and Woodward, 1956). In the connected lakes, small centric diatoms, and small chlorophytes such as *Scenedesmus*, *Ankyra* and *Monoraphidium* were common members of the phytoplankton community. Since they are generally considered more edible for zooplankton, the top-down control of phytoplankton may have been reinforced by the short WRT (e.g. Timms and Moss, 1984) during the spring. Short WRTs may favour rapidly reproducing species that do not form large colonies and are therefore more easily grazed by zooplankton (Brook and Woodward, 1956; Dickman, 1969).

The physical effects of River Erewash may have been augmented by the supply of nutrients to the connected lakes. Jensen et al. (1994) argued that in hypertrophic temperate shallow lakes, chlorophytes were more dominant than cyanobacteria because of their greater requirement for nutrients to satisfy their rapid growth rates. The importance of the River Erewash for supplying N to the connected lakes may also contribute to the suppression of heterocystous cyanobacteria. This idea is explored in

more detail in section 7.3.3, page 347. It is difficult to accurately attribute the increase in the importance of chlorophytes to either the effect of flushing rates or nutrient supply.

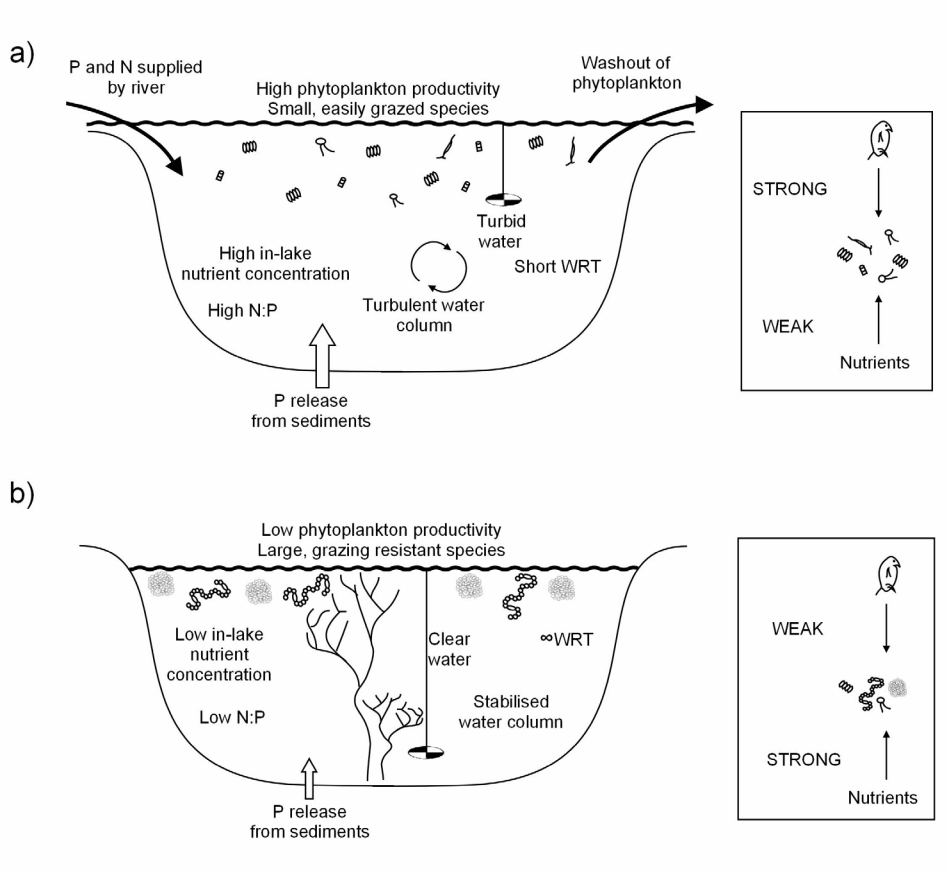

Figure 7.4 Potential mechanisms which may favour: a, high phytoplankton biomass and the growth of small, easily grazed species (such as chlorophytes and diatoms) in lakes connected to nutrient-rich rivers; and b, lower phytoplankton biomass and the growth of large, grazing resistant phytoplankton species (e.g. cyanobacteria) in lakes isolated from rivers. The potential strength of top-down and bottom-up processes during spring are shown on the right of the diagram.

In lakes I1 and I2, zooplankton grazing was weakly negatively associated with total phytoplankton biomass (r_s ≤0.543, see Table 4.15, page 219). This suggests that top-down control of the zooplankton may be moderated by the higher proportions of grazing-resistant cyanobacteria in the phytoplankton of the isolated lakes during the summer compared to the connected lakes (Figure 7.5). Cyanobacteria are likely to be less easily grazed due to their size (Agrawal, 1998). Zooplankton may also be

sensitive to the toxicity of cyanobacteria (Haney, 1987). Another possible competitive advantage for cyanobacteria in the connected lakes may be their regulation of buoyancy, which may help to avoid sinking losses in a water-column stabilised by submerged macrophytes (Reynolds et al., 1987).

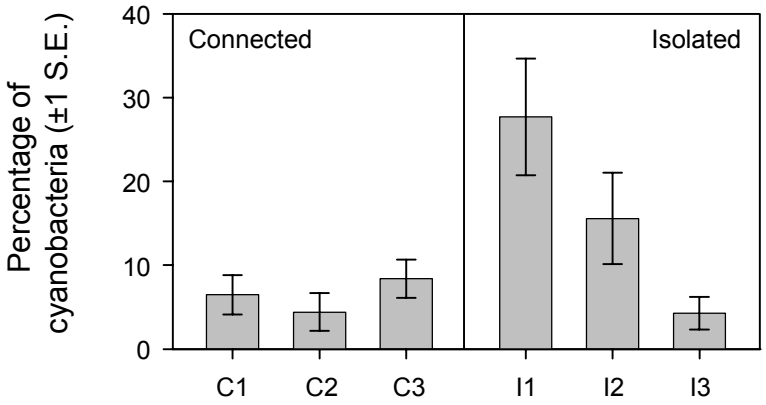

Figure 7.5 Mean percentage (±1 S.E.) of cyanobacteria in the phytoplankton of the lakes of Attenborough Nature Reserve during the growth seasons (June-September) of the monitoring period.

Bottom-up control probably also influenced the phytoplankton communities of isolated lakes more strongly than in the connected lakes. However, strong negative correlations were found between chlorophyll-*a* and NH_4-N, NO_3-N and SiO_3 concentrations (Chapter 4, Table 4.15) in the connected lakes suggesting high uptake of nutrients associated with the high phytoplankton biomass during summer. NH_4-N depletion during the summer has been observed in other eutrophic lakes (e.g. Lau and Lane, 2002; Présing et al., 2008). NH_4-N limitation in the connected lakes may have been alleviated by uptake of NO_3-N when NH_4-N supply was exhausted (Dortch, 1990). Negative correlation between SiO_3 concentrations and chlorophyll-*a* is likely to occur where diatoms form a significant proportion of the planktonic phytoplankton biomass. In connected lakes of Attenborough Nature Reserve, diatoms formed more than 31% of the phytoplankton community during the monitoring period.

In isolated lakes I1 and I2 NO_3-N was significantly related to phytoplankton community structure in both lakes I1 and I2 (Table 4.15, page 219). Competition for NO_3-N with submerged macrophytes (e.g. Ozimek et al., 1990) may have reduced the availability of NO_3-N for phytoplankton, particularly during the summer, and may have favoured

cyanobacteria. NH_4-N, SiO_3 and SRP were also significantly associated with phytoplankton community structure in Lake I1, which may indicate a greater strength of bottom up control of phytoplankton than in Lake I2. In Lake I3, both NH_4-N and SRP were significantly related to phytoplankton community biomass and structure (Table 4.15, page 219), suggesting that bottom-up control of phytoplankton was stronger than top-down controls.

However, the traditional view that cyanobacteria become more dominant at high nutrient concentrations may not be applicable where nutrient-rich river inflows decrease WRT and increase nutrient concentrations to a hypertrophic state. Connectivity to such rivers may, paradoxically, be advantageous for the management of eutrophic shallow lakes by favouring a phytoplankton community that is more easily controlled by zooplankton grazing. However, predation on zooplankton by fish may reduce the strength of top-down control on phytoplankton by favouring smaller and more inefficient grazers (Brooks and Dodson, 1965). This has an important implication for lake restoration as it suggests that increasing the biomass of zooplankton (for example by biomanipulation) has more potential for reducing phytoplankton abundance in hypertrophic shallow lakes where cyanobacteria are less abundant, than lakes with lower nutrient concentrations, proportionally abundant cyanobacteria and long WRTs.

7.2 Investigate how flooding events may affect the biology and chemistry of shallow lakes

7.2.1 Summary of major findings

- Flooding was associated with lower P concentrations;
- N and Si concentrations did not change during flooding;
- Flooding was associated with reduced phytoplankton abundance;
- Flooding favoured cryptophytes and diatoms and reduced the abundance of cyanobacteria;
- The effects of short WRT associated with connectivity to rivers might be advantageous for shallow lake management.

7.2.2 Nutrient concentrations

P concentrations during the wet year of 2007 were lower than during 2005 and 2006 in both connected and isolated lakes. This was probably associated with dilution of P in the inflowing River Erewash and washout through outflows in the connected lakes (Figure 7.6) and dilution by increased water volumes alone in lakes I1 and I2. This is important for the potential of river diversion as a technique for lake restoration, as it suggests that maintaining connectivity to inflows and outflows may help to reduce in-lake P concentrations. A management approach that maintains flushing rates contrasts with the River Erewash diversion strategy currently being persued at Attenborough Nature Reserve.

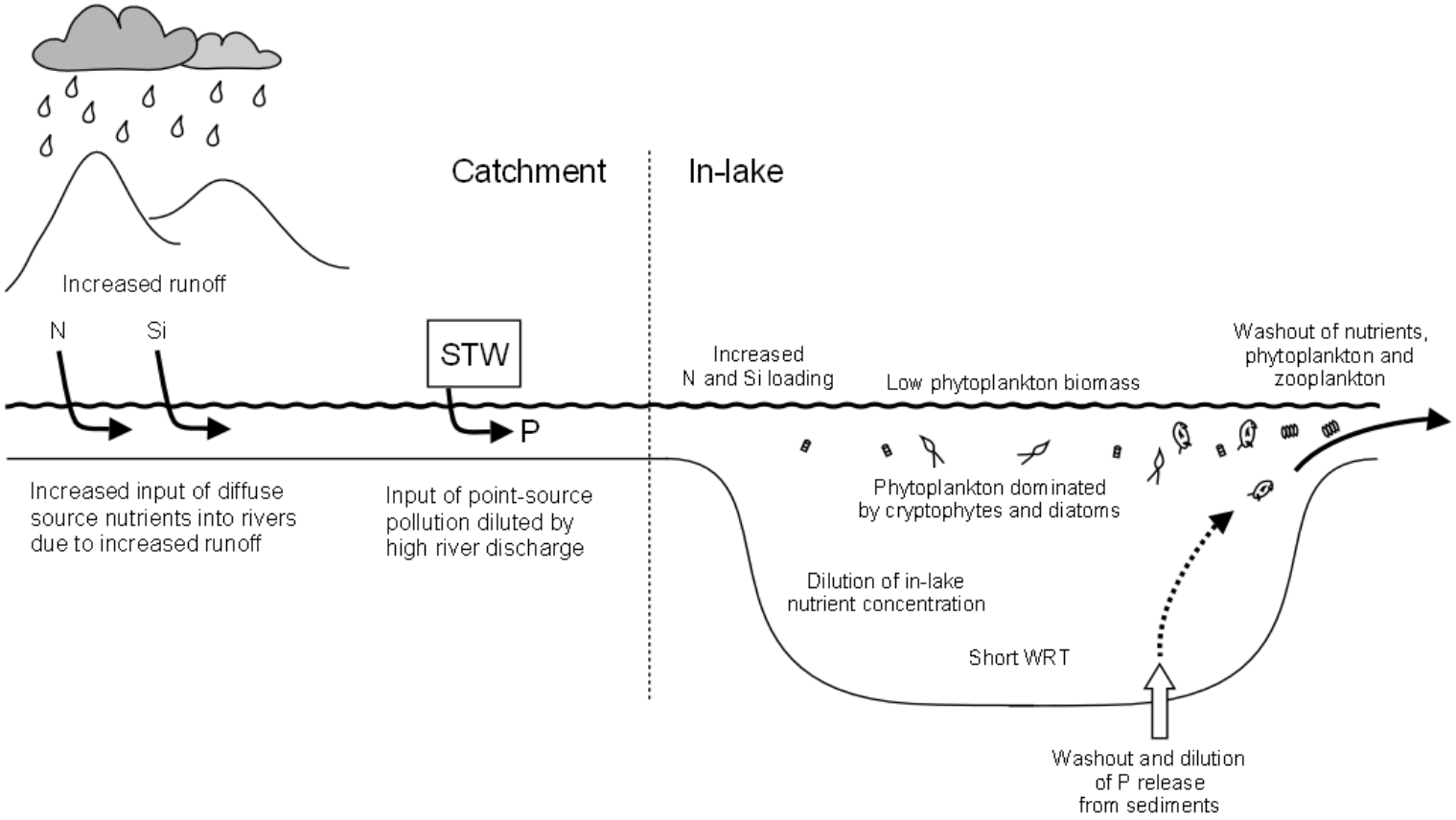

Figure 7.6 Changes in the connected lakes associated with increased precipitation.

Management of discharge from the lakes to decrease WRT during summer may allow for P released from the sediments to be flushed from the lake in order to reduce water-column P concentrations. A management strategy which regulates the flushing rates of Attenborough Nature Reserve could be particularly advantageous because of the high P release from the sediments. Spears et al. (2006) argue that as many lakes that are impacted by eutrophication have managed hydrological regimes, flow management could be widely used to aid lake recovery. The flow of water through Attenborough Nature Reserve could potentially be easily manipulated using existing flow management structures, such as the weir at MPO and flaps on Ea outflow. Flushing with nutrient-poor water during winter has been suggested to reduce cyanobacterial blooms (Hosper, 1998), although the success of this winter flushing was hampered by sedimentary P release during summer. Increasing flushing during the summer has the advantage of directly removing phytoplankton and reducing the effects of summertime P release, as shown by the differences in biology and nutrient concentrations observed during the wet year of 2007 (section 5.6, page 260 and section 5.7, page 265).

The in-lake concentration of N and Si did not vary much during the wet year of 2007 compared to 2005 and 2006. As both N and Si are primarily derived from diffuse sources (Petzoldt and Uhlmann, 2006; Sferratore et al., 2006), and the transport of nutrients derived from diffuse sources increases when the discharge into lakes increases (Elliott et al., 2009), dilution and washout are likely to have balanced the increase in N and Si transport into the connected lakes. This caused increases in the DIN:SRP and Si:SRP ratios (Figure 7.7). This is potentially significant for the management of shallow lakes connected to nutrient-rich rivers as it suggests that future variations in discharge may have complex effects on in-lake nutrient concentrations. Increased discharge may result in lower P concentrations and increases in the N:P and Si:P ratios. Reduced discharge may have the opposite effect, because the retention of P in the water column associated with reduced flushing rates may further decrease N:P and Si:P ratios, potentially promoting cyanobacterial blooms.

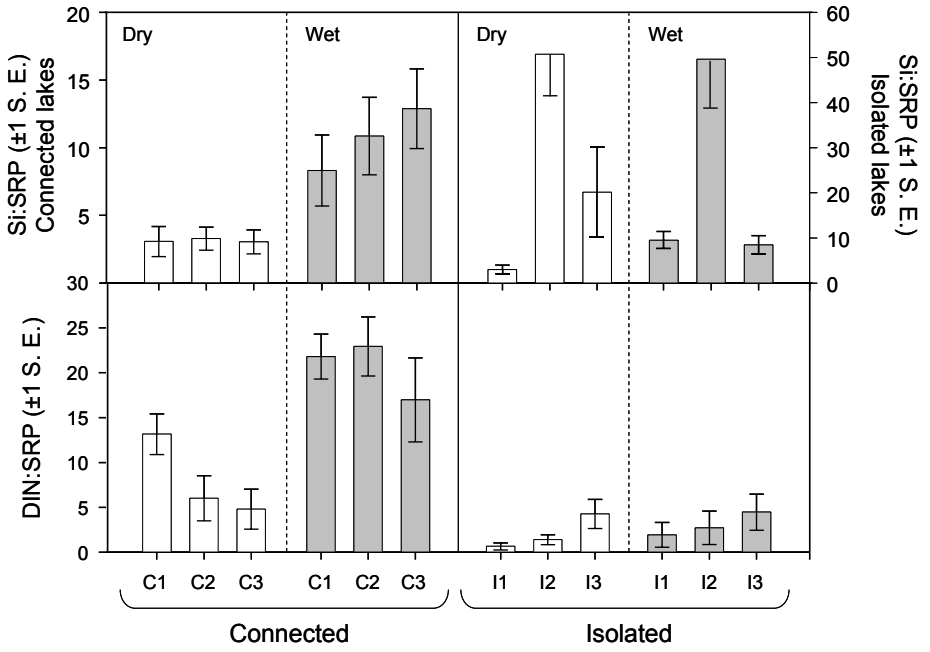

Figure 7.7 Mean molar DIN:SRP and Si:SRP ratios (±1 S.E.) in connected and isolated lakes during dry (2005–2006) and wet (2007) growth seasons (June–September). Note change in *y*-axis for isolated lakes.

7.2.3 Phytoplankton abundance and composition

The effects of reduced WRTs on phytoplankton biomass was desirable for lake management and restoration as low WRTs were associated with decreased biomass (Figure 7.6). Achieving a decrease in the total biomass of phytoplankton is frequently an important part of lake restoration attempts in order to reduce turbidity. However Secchi disk depths during 2007 in the connected lakes were only slightly (~10–15 cm) greater than during 2005 or 2006, suggesting that minerogenic turbidity was probably important during 2007. Short WRTs during the summer may therefore have desirable effects on the phytoplankton of lakes, but may not reduce water turbidity. Diverting rivers from lakes is likely to prevent episodic flooding events from diluting and washing-out phytoplankton.

The changes in the phytoplankton community associated with reductions in WRT were also complementary to the aims of lake restoration. Cyanobacterial biomass was reduced and diatoms and cryptophytes became more important (Figure 7.6). These changes are also advantageous to lake managers as the likelihood of cyanobacteria

blooms was reduced. These findings suggest that lake restoration that maintains short WRTs may be a more successful strategy than isolation. The lasting effects of the flooding during 2007 are unknown from the duration of this study. Therefore, future studies should monitor the recovery of lakes from episodic flooding to establish if short-term reductions in flushing rate have advantages for lake restoration and management or whether sustained reductions are required to reduce phytoplankton abundance and composition.

7.3 Investigate the effects of nitrogen and silica on ecosystem community structure

7.3.1 Summary of major findings

- N did not have any significant effect on total phytoplankton biomass;
- Low N:P ratios are associated with higher biovolumes and proportions of cyanobacteria in the phytoplankton;
- Si did not have any significant effect on phytoplankton biomass or composition;
- Si:P ratios are a poor predictor of absolute and relative diatom abundance in Attenborough Nature Reserve.

7.3.2 Effects of N on total phytoplankton biomass

The evidence presented in this thesis cannot conclusively confirm the idea that increases in N concentrations increase the biomass of phytoplankton in eutrophic shallow lakes. The results of the mesocosm experiment were inconclusive, so the effects of N can only be assessed at a whole-lake scale. A strong positive correlation between the mean chlorophyll-*a* concentration and mean DIN measured during the monitoring period (r_s = 0.886, p = <0.02) in all of the lakes suggests that higher mean DIN concentrations are associated with higher mean chlorophyll-*a* concentrations. NH_4-N was closely associated with phytoplankton biomass, and negative correlations

were found between NH_4-N and chlorophyll-*a* concentrations in the connected lakes, in common with other studies (e.g. Lau and Lane, 2002). In isolated lakes I1 and I2 the very low N concentrations and presence of N-fixing cyanobacteria (Chapter 3) suggests that N may have been limiting during the summer. NH_4-N concentrations may have been associated with the non-significant response of the mesocosm experiment to N enrichment (Chapter 6). The apparent importance of NH_4-N for phytoplankton suggests that diverting nutrient-rich rivers may result in total phytoplankton biomass being controlled by the rates of internal regeneration of NH_4-N. Mechanisms of internal NH_4-N regeneration that may be of significance to Attenborough Nature Reserve include release from the sediments (e.g. van Luijn et al., 1999; Beutel, 2006) and zooplankton excretion (e.g. Wen and Peters, 1994; Attayde and Hansson, 1999; Vanni, 2002). Losses of NH_4-N to denitrification may further enhance NH_4-N limitation (Présing et al., 2008). Future studies could consider the NH_4-N release from the sediments of connected lakes so that potential for NH_4-N to become limiting after the diversion of the River Erewash can be assessed. It is possible that reducing both NH_4-N and P supplies to eutrophic lakes may increase the likelihood of cyanobacterial blooms (Présing et al., 2008).

7.3.3 N:P ratios and cyanobacteria

In Attenborough Nature Reserve, the data tentatively support the hypothesis that low N:P ratios may favour cyanobacteria. Cyanobacteria were more abundant in Lakes I1 and I2 (Figure 7.5), and growth season mean DIN:SRP ratios were lowest in the isolated lakes (Figure 7.7). The strength of the correlation between cyanobacterial abundance and DIN:SRP ratios appeared to be slightly stronger in the isolated lakes than the connected lakes (Figure 7.8). In the connected lakes, cyanobacteria were most abundant when the DIN:SRP ratio was <5, and in connected lakes when DIN:SRP was between 5.0 and 32. The different strength of the effects of DIN:SRP ratios on the relative and absolute biovolume of cyanobacteria between connected and isolated lakes suggests that WRT may reduce the effect of DIN:SRP ratios. In the connected lakes, flushing may have reduced the abundance of cyanobacteria (see Figure 7.4). Additionally, nutrient ratios may not accurately predict phytoplankton response if either nutrient is non-limiting (Reynolds, 1984) which may reduce the predictive strength of DIN:SRP in connected lakes where N rarely became limiting. For

most of the growth season, NO_3-N concentrations in the connected lakes were in excess of the 0.14 mg L^{-1} of NO_3-N that Présing et al. (1997) suggested may be responsible for reducing the abundance of N-fixing cyanobacteria. Dodds (2003) suggests that DIN:SRP ratios are not the most accurate measures of nutrient availability and limitation. However, because DIN and SRP is easier to measure than total-N and total-P, if managing cyanobacteria is a management priority for shallow lakes, then the DIN:SRP ratio may be a cost-effective and easily-obtainable measurement for predicting the dominance of cyanobacteria compared to TN:TP ratios, particularly if ambient N concentrations are generally low.

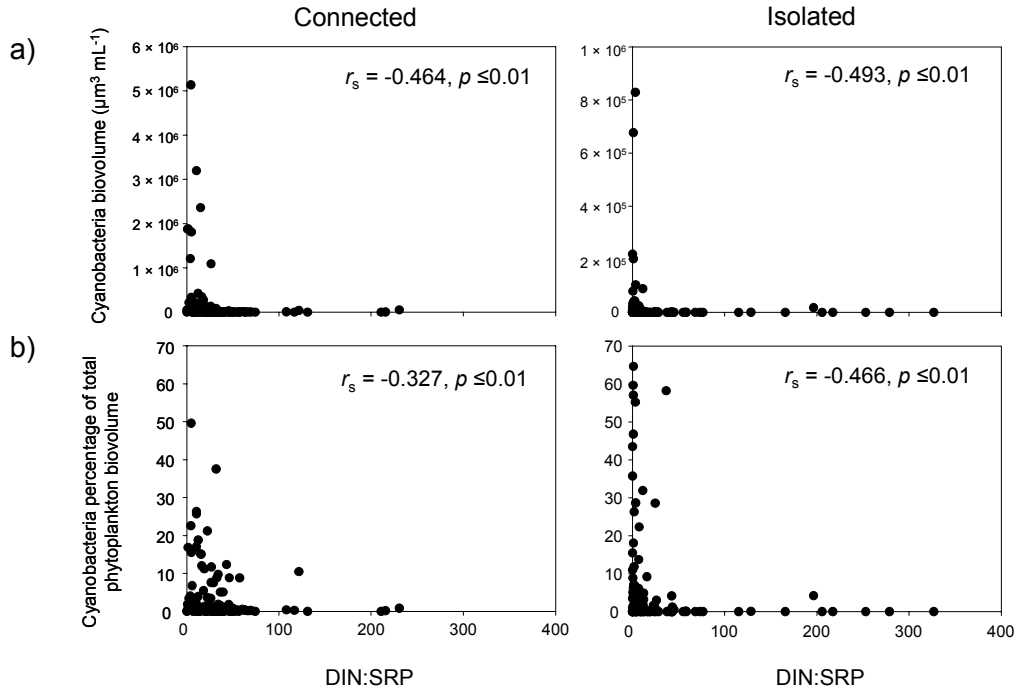

Figure 7.8 The relationship between the DIN:SRP ratio (by molarity) and a) the absolute biovolume of cyanobacteria; b) the proportion of total phytoplankton biovolume comprising cyanobacteria. Spearman's rank correlation was used to assess the strength of the relationship.

Cyanobacteria were rare (Figure 7.4) in Lake I3, even though the DIN:SRP ratio was low. Lake I3 is therefore a clear exception to the hypothesis that low N:P favour cyanobacteria. Given that P is more abundant than in Lake I3 than in the other isolated lakes, conditions appeared to be ideal for dominance by cyanobacteria. One possible explanation is exception that the high Si:P ratio may have favoured the

growth of diatoms over cyanobacteria (Holm and Armstrong, 1981; Horn and Uhlmann, 1995) although the likelihood of this being important is not high (see below). Présing et al. (1997) suggested that cyanobacteria were absent from a hypertrophic lake where the N:P ratio was low but, NH_4-N abundant. This may be analogous to the findings of the mesocosm experiment where NH_4-N was apparently high enough to prevent cyanobacteria growth, despite very low NO_3-N concentrations and an abundance of P. N:P ratios may accurately help predict patterns of cyanobacteria distribution at landscape scales (e.g. Patoine et al., 2006), but within individual lakes may be of less predictive power because of the confounding effects of the relative availability of NO_3-N and NH_4-N.

7.3.4 Effects of Si on total phytoplankton biomass

Diatoms were a significant component of the phytoplankton community of the connected lakes and Lake I3, and were probably responsible for the depletion of Si during the spring and early summer. Comparing the mean concentration of Si between lakes with the mean biovolume of diatoms (Figure 7.9) shows that among lakes, higher Si concentrations favour greater biovolumes of diatoms. Chapter 4 suggested that total phytoplankton biomass was strongly negatively associated with Si concentration. Negative correlations also exist between diatom biovolume and Si concentrations (r_s = -0.506 – -0.605, $p \leq 0.01$) in the connected lakes and Lake I3 (r_s = -0.407, $p \leq 0.05$), probably reflecting uptake of Si by diatoms. No significant correlation between diatom biovolume and Si concentration was found in Lakes I1 and I2. The strength of negative diatom-Si correlation may have been higher in the connected lakes compared to isolated lakes I1 and I2 because the short WRT may have favoured small phytoplankton that are capable of rapid reproduction and benefit from more turbulent water columns (Figure 7.4a). Many diatoms are classified as *R*- strategists and may be prone to sinking (Reynolds et al., 1982; Poister and Armstrong, 2003; Reynolds, 2006) and so thrive in the connected lakes. The diversion of the River Erewash may reduce the role of Si in determining total phytoplankton biomass by increasing the proportion of non-siliceous species in the phytoplankton community (Figure 7.4). Experimental data (Chapter 6) did not provide any further insights into the role of Si in shallow lakes. Diatom biomass did not significantly increase where Si was added. Zooplankton grazing and sinking may have been related to the scarcity of

diatoms in the mesocosms during the experiment. Lower chlorophyll-*a* concentrations in the +Si treatments may have been associated with the higher abundance of *Ceriodaphnia* sp. and *Bosmina* spp. in the +Si mesocosms (section 6.4).

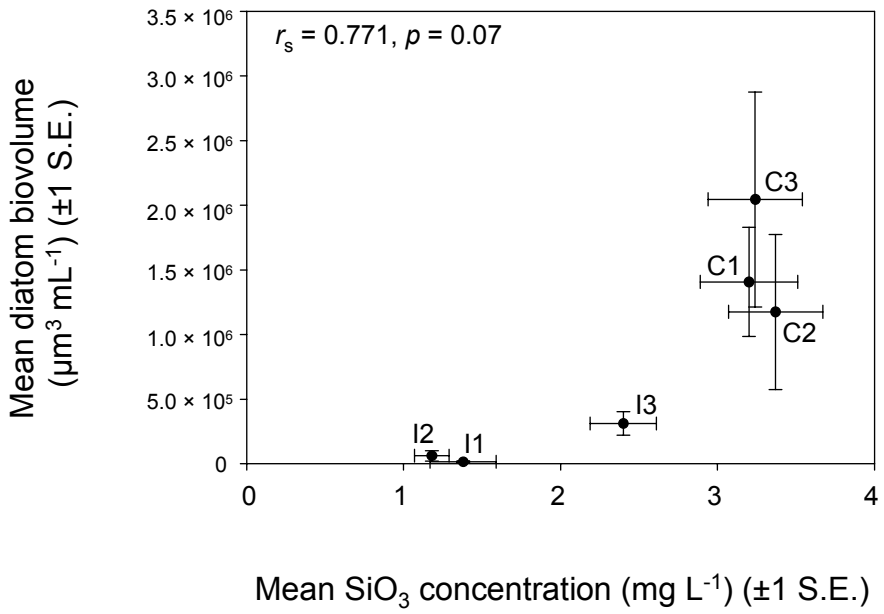

Figure 7.9 Mean SiO_3 concentrations and mean diatom abundance (± 1 S.E.) in Attenborough Nature Reserve, Mach 2005 to March 2008.

7.3.5 Si:P ratios and diatoms

Diatoms may have a competitive advantage over other phytoplankton when Si:P ratios are high (see section 1.4.3, page 47; Tilman et al., 1986). However, the data collected from Attenborough Nature Reserve do not support this, despite the strength of correlation between mean Si concentration and mean diatom biovolume. Weak negative correlations were found between the Si:P ratio, and the biovolume of diatoms or their relative abundance over time (Figure 7.10). The highest biovolumes and proportional composition of diatoms generally occurred at the lowest Si:P ratios. Mean Si:P ratios during the monitoring period in each lake were not significantly associated with either mean diatom biovolume (r_s = -0.765, p = 0.08) or the percentage of total phytoplankton biovolume as diatoms (r_s = -0.530, p = 0.28). These findings are probably principally associated with the uptake of Si by diatoms (Bailey-Watts, 1976),

which depletes water column concentrations of Si more strongly than for P, and therefore reduces the Si:P ratio.

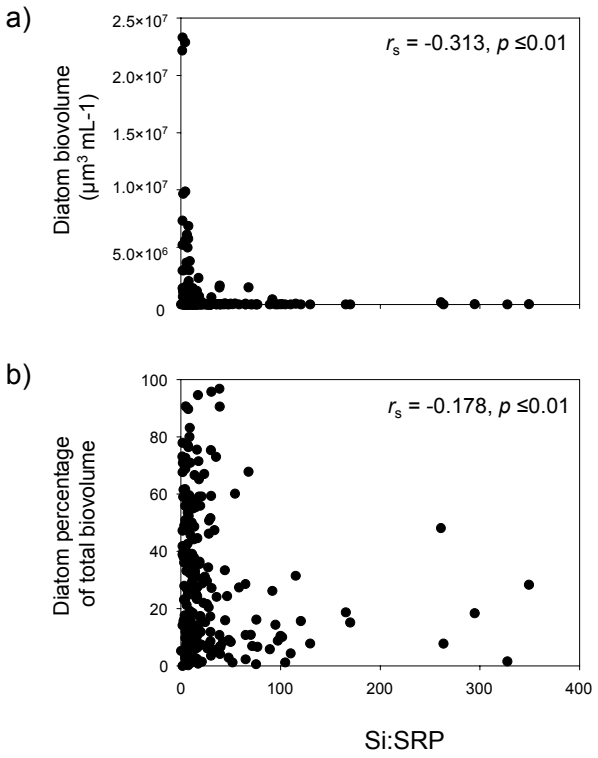

Figure 7.10 The relationship between the Si:SRP ratio (by weight) and a) the absolute biovolume of diatoms; b, the proportion of total phytoplankton biovolume comprising diatoms. Spearman's rank correlation (r_s) was used to assess the strength of the relationship.

7.4 Explore the likely effects of diverting the River Erewash from Attenborough Nature Reserve

7.4.1 Summary of major findings

- Internal loading of P may delay lake responses to river diversion;
- In-lake N concentrations respond rapidly to reductions in N loading, but because internal loading maintains high P concentrations, N:P ratios may fall and favour cyanobacteria after river diversion;

- Submerged vegetation is unlikely to develop rapidly, because the mechanisms reinforcing the turbid state are not removed by river diversion alone;
- Additional measures are likely to be required to promote a switch to a stable clear-water state in the short term.

7.4.2 Delayed response of in-lake P concentrations

Data presented in this thesis have suggested that internal P release is an important mechanism that increases the late-summer in-lake P concentration of the connected lakes (Chapter 3). The mesocosm experiment also provided evidence of the importance of P release (Chapter 6). Therefore, the response of water-column P concentrations is likely to be delayed until internal loading declines. P concentrations in shallow lakes may continue to decline for 10-15 years after external loading reductions (Jeppesen et al., 2005b; Phillips et al., 2005). Historically high P loadings and high WRTs may delay reductions of P concentration response (Jeppesen et al., 2005b). The response of spring and early summer P concentrations to loading reductions are often quicker than for late summer P concentrations (Köhler et al., 2005; Phillips et al., 2005; Søndergaard et al., 2005). Isolation may prevent the washout and dilution of P released from the sediments, but also decreased water level may concentrate internal P loading (Perrow et al., 1994).

The implications for the use of river diversion as a technique for lake restoration is that internal loading of P is likely to maintain pre-diversion P concentrations during the late summer in Attenborough Nature Reserve because of internal P release, even if decreases are possible during the early summer. This may mean that the phytoplankton biomass of the connected lakes in the Attenborough Nature Reserve would decrease early summer (e.g. Phillips et al., 2005) and remain at pre-diversion quantities in late summer. Connected lakes may become strongly N limited as in lakes I1 and I2. Moss et al. (1986) found that isolation reduced chlorophyll-*a* concentrations for four years after isolation, but this was probably associated with strong N limitation and the growth of submerged macrophytes.

7.4.3 Increases in abundance of cyanobacteria

Diverting nutrient-rich rivers may be a particularly efficient technique for reducing in-lake N concentrations (see section 7.1.2, page 331). N is often derived from diffuse rather than point sources so can be difficult to control in comparison to P unless catchment-wide measures are implemented. River diversion therefore substantially reduces in-lake concentrations of N (Moss et al., 1986; Perrow et al., 1994). Regardless of the nutrient reduction technique used, reductions in external N loading frequently result in rapid in-lake responses (e.g. Jeppesen et al., 2005b; Köhler et al., 2005). In most lake restoration measures, increases in the TN:TP (and DIN:SRP) ratio occur, because P is the nutrient targeted for reduction, although river diversions will probably decrease this ratio because of isolation from N loading and continued internal P loading. As has been discussed, low N:P ratios in Attenborough Nature Reserve are associated with a greater proportion and biovolume of cyanobacteria in the phytoplankton community (see section 7.3.3, Figure 7.5) and it may be expected that a decrease in N:P ratios caused by diverting the River Erewash may favour their growth. This has been observed in other shallow lakes. In Alderfen Broad, Norfolk, the cyanobacterium *Anabaena* became an abundant component of the phytoplankton six years after isolation (Moss et al., 1986; Perrow et al., 1994), and a shift towards *Anabaena* and a decrease in DIN:SRP ratios to <1 occurred simultaneously in Mügelsee (Köhler et al., 2005). The increase in WRT associated with the diversion of rivers may further favour cyanobacteria (Figure 7.4b). Cyanobacteria may present a significant management problem because few techniques are available to control them. One technique that may help manage the risk of large cyanobacteria blooms is the addition of barley straw into lakes. The release of substances toxic to phytoplankton by decaying barley straw has been shown to successfully reduce the abundance of cyanobacteria and other phytoplankton in lakes and reservoirs with no observed side effects on other aquatic organisms (e.g. Everall and Lees, 1996; Barrett et al., 1999; Brownlee et al., 2003).

7.4.4 Delayed recovery of submerged vegetation

It is unlikely that the diversion of the River Erewash alone will result in a rapid recovery of submerged vegetation in the short-term in the connected lakes, even if NO_3-N

concentrations will probably be sufficiently low for diverse and stable macrophyte communities (<1–2 mg L^{-1}, González Sagrario et al., 2005; James et al., 2005). The principal reason for this is because the feedback mechanisms which stabilise shallow lakes in a turbid–water state will not be overcome (Lauridsen et al., 2003; Hilt et al., 2006). As the previous sections have suggested, continued internal loading of P will probably maintain a high biomass of phytoplankton, preventing sufficiently clear water for macrophyte establishment.

On the assumption that internal P loading will eventually decline, and water clarity improve for the potential for submerged macrophyte growth, there remains no guarantee that submerged macrophytes may establish a stable community. After isolation from a nutrient-rich inflow, Alderfen Broad, Norfolk, rapidly developed an abundant community of the submerged macrophyte *Ceratophyllum demersum* (Moss et al., 1986; Moss et al., 1990) although this disappeared after seven years from the isolation and phytoplankton became dominant again. This may reflect the sedimentation of organic plant matter eventually causing large quantities of P release from the sediments, or a build-up of substances that could prevent plant growth, such as sulphides (Moss et al., 1990). The lack of flushing to remove these substances may have been particularly important (Moss et al., 1990), and emphasises the importance of lake flushing. This suggests that the re-establishment of a stable macrophyte community may be a particular challenge in the restoration of Attenborough Nature Reserve.

The presence of viable propagules in the sediments is of importance for the development of a submerged macrophyte community. Charophytes are particularly useful in shallow lake restoration because of their strong effects on water clarity (Van den Berg et al., 1998a; 1998b). A viable propagule bank has been shown to be particularly important for the establishment of *Chara* spp. (Van den Berg et al., 2001). Viable propagules of aquatic macrophytes have been found at depth in lake sediments, although this depends on the previous presence of submerged macrophytes (Amano et al., 2008). The presence of viable propagules in the sediments of the connected lakes is currently unknown. Investigating this could be of value for developing an appropriate management strategy for establishing submerged vegetation. If insufficient propagules exist for re-establishing vegetation, then the

artificial introduction of submerged macrophytes may be required. Lakes I1 and I2 are potentially a cost-effective source for macrophytes for this purpose. The abundant bird population of Attenborough Nature Reserve may transport macrophyte propagules naturally between lakes although this mechanism should probably not be relied on to establish a macrophyte community. Emergent macrophytes may provide a refuge for zooplankton (Cazzanelli et al., 2008) and reduce sediment resuspension (Horppila and Nurminen, 2001) which may be a useful interim measure before submerged macrophytes become established. Grazing by birds, particularly coot (*Fulica atra*), may hinder the growth of submerged macrophytes (e.g. Lauridsen et al., 1993) although the significance of this remains unresolved (Perrow et al., 1997b; Hilt, 2006). Disturbance and grazing by fish may also hamper the development of submerged macrophyte communities (De Winton et al., 2002; Körner and Dugdale, 2003; Nurminen et al., 2003). Both fish and birds may prove a significant hindrance to the development of submerged macrophytes in the connected lakes, because of the large waterfowl populations present at the Attenborough Nature Reserve and probable high abundance of benthivorous fish (including carp, bream and roach). Enclosure cages may be useful for protecting macrophytes from birds and fish if macrophyte communities develop (Hilt et al., 2006).

7.4.5 Techniques for complementing the River Erewash diversion

The discussion above has suggested that the diversion of the River Erewash alone is unlikely to be sufficient to cause a switch from the turbid-water state to a stable and clear-water state in the short term (i.e. <10 years). Evidence from other lake restorations supports this (Lauridsen et al., 2003; Jeppesen et al., 2005b), suggesting little increase in submerged macrophyte abundance after P-loading reduction. However, over the long-term, submerged vegetation may recolonise naturally, as appears to have happened in Lake I2. Lake I2 was isolated from the River Erewash in 1981. Improvements in light penetration and a reduction in nutrient loading were probably the main effects that contributed towards the development of a stable macrophyte community. Although the decline in nutrient concentrations in Lake I2 associated with isolation from the River Erewash is unknown, a reduction in NO_3-N

concentration may have been important for the recolonisation of submerged macrophytes. One potential option for the restoration of the connected lakes would be to do nothing in addition to river diversion, in the assumption that natural processes would complete the restoration (Bradshaw, 1996). Natural processes that may of importance include the gradual change towards equilibrium P concentrations and increases in submerged macrophytes. The advantages of using natural processes in restoration include their low cost, probable self-sustaining nature and suitability for large-scale use (Bradshaw, 1996). Allowing natural processes to restore the connected lakes may eventually lead to their complete restoration. Meeting the aims of the river diversion (reducing P concentrations to 100 μg L^{-1} and reducing 'excessive' phytoplankton biomass, White Young & Green, 2006) are unlikely to be achieved for 10-15 years unless measures are undertaken to aid the development of submerged vegetation. This section presents and discusses ideas that could be used after river diversion to aid the short-term development of submerged vegetation in Attenborough Nature Reserve. These techniques are summarised in Figure 7.11.

Division of connected lakes

Isolating sections of lakes C1, C2 and C3 may be useful in order to create areas where conditions favourable for submerged macrophytes could be created that could in future colonise the remainder of the lakes (Qiu et al., 2001). The complex geography of the connected lakes would allow for sections of the lakes to be separated and isolated from the rest of the lakes easily. Sediment removal and biomanipulation could be used in these isolated areas to improve the stability of the vegetation communities. This approach would reduce the initial costs of restoration due to the smaller areas where expensive techniques would be used, and prevent conflict with the passage of gravel barges through the Reserve. The creation of areas rich in submerged macrophytes would also be beneficial for the conservation and amenity value of Attenborough Nature Reserve. Enclosures could be removed when water clarity in the remainder of the lakes is suitable for submerged macrophytes. Experiences from the Norfolk Broads have demonstrated the feasibility of isolation of sections of lakes. Installing plastic sheeting or solid wooden walls are possible techniques for isolation, both resulting in the complete isolation of the water from the rest of the lake, which would increase the efficiency of sediment removal, and the prevention of the movement of fish across the barrier (Moss, 2001).

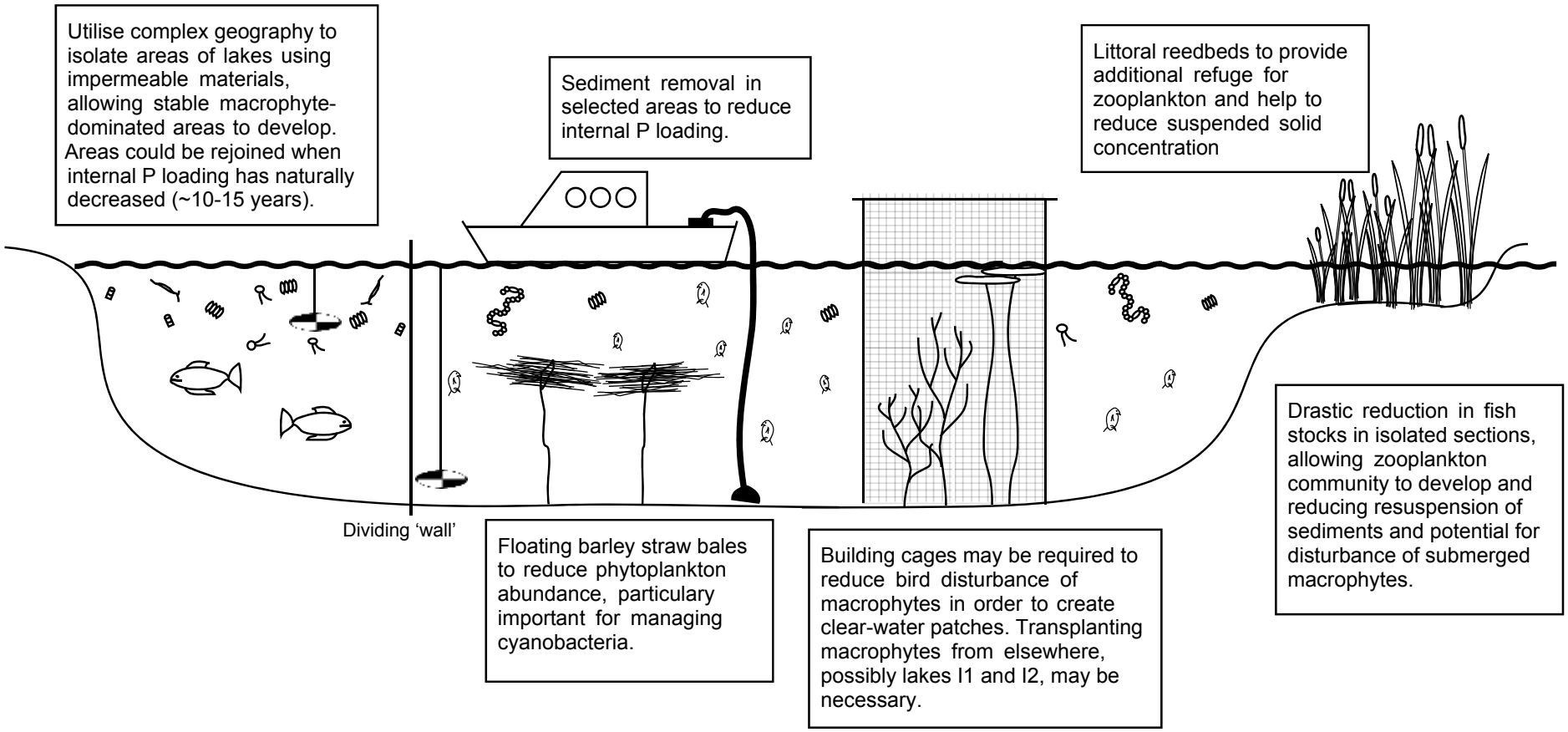

Figure 7.11 Potential techniques that could be used in Attenborough Nature Reserve after the diversion of the River Erewash to improve the probability of developing stable dominance by submerged vegetation and clear water. Secchi disks indicate relative expected turbidity

Littoral reed beds

Littoral reed beds could be established both within the enclosures and outside of them. Reed beds can help to reduce suspended solid concentrations and may provide a refuge for zooplankton which could help reduce phytoplankton abundance (Horppila and Nurminen, 2001; Cazzanelli et al., 2008). Reed beds are aesthetically attractive and could be established relatively easily in areas where littoral water depths are shallow. Reedbeds are currently found throughout Attenborough Nature Reserve, and additional reedbed habitat is currently being created in Lake C1 as part of mitigation for the flood alleviation scheme in the reserve (Broxtowe Borough Council, 2009). Increasing the area of reedbeds would benefit the variety of wildfowl that contribute to the site's classification as a SSSI, including reed buntings (*Emberiza schoeniclus*), reed warblers (*Acrocephalus scirpaceus*) and sedge warblers (*A. schoenobaenus*, Natural England, 1981), in addition to potentially improving water clarity.

Reducing internal P loading

In lakes that have received high P loadings for several years, the removal of sediments may help to reduce water-column P concentrations (Phillips et al., 1999). In Cockshoot Broad, Norfolk, 70 cm of sediment was removed after the lake was isolated from nutrient-rich river inflow (Moss et al., 1986; 1996b). Total P and chlorophyll-*a* concentrations declined, and submerged macrophytes and *Daphnia* increased immediately after the sediment removal (Moss et al., 1986; 1996b). However, two years later, the P and chlorophyll-*a* concentrations increased again, which may have been related to input to the sediments of large amounts of organic matter from submerged vegetation. Unless all sediment is removed, P retained at depth can diffuse through the anaerobic sediments into the water column (Moss, 2001). Sediment removal is very expensive (Kleeberg and Kozerski, 1997; Annadotter et al., 1999), and requires careful handling of the dredged material to prevent the return of nutrients into lakes and rivers (Moss et al., 1996a). Restricting sediment removal to targeted sections of the lakes would help control the cost of dredging and reduce the volume of material to be handled.

Suppressing algal growth

The addition of barley straw may be cost effective way of reducing phytoplankton abundance after the diversion of the River Erewash whilst in-lake nutrient concentrations remain high. Barley straw could be a cost effective method in mitigating against the likelihood of cyanobacterial blooms developing after river diversion because of the reduction in flushing rate and N:P ratio. Barrett et al. (1999) found that distributing small floating bales of straw around a lake was logistically the easiest method, and allowed for the straw to decompose, which is important for the effectiveness of the method. It would be possible to place hay bales away from littoral areas in the Attenborough Nature Reserve to avoid any negative aesthetic impacts and interference with angling. Care would need to be taken to avoid conflict with the passage of barges through the connected lakes.

Reducing predation on zooplankton

Manipulation of food webs is often used in shallow lake restoration after the reduction of nutrient loading. Compared to chemical methods, biomanipulation is relatively cheap (Jeppesen et al., 2007). The principle aim of biomanipulation is to increase the abundance of large-bodied zooplankton, in order to increase top-down control of the phytoplankton and improve water clarity (Figure 7.12). Biomanipulation can result in rapid reductions in the phytoplankton biomass in a relatively short period (Søndergaard et al., 2008). The mesocosm experiment demonstrated that the zooplankton community of Lake C2 rapidly increased after protection from predators (section 6.3.3, page 314), and was probably responsible for the reduction in chlorophyll-*a* concentration in the mesocosms. This suggests that biomanipulation is a potentially effective technique to augment responses to a reduction in external nutrient loading in Attenborough Nature Reserve. However, conflict with angling interests in the Reserve may be problematic unless fish manipulations are restricted to isolated areas only.

Biomanipulation is typically done by removing a large proportion of the fish from a lake, and returning only piscivorous species, such as pike, in order to remove the small remaining number of zooplanktivorous species. Periodically repeating fish manipulation measures to maintain the desired biomass is usually required because of recruitment by the remaining fish, enhanced by increased food availability (Gulati et

al., 2008). The addition of piscivorous species (e.g. pike, *Esox lucius*) is an alternative to fish removal (Moss, 2001), although few successes of using this technique are known (Skov et al., 2007; Søndergaard et al., 2007). One of the largest concerns of using biomanipulation in shallow lakes is the stability of the clear-water state. In many cases, turbid water returns after 5-10 years, which may be associated with insufficient reduction in P loading (either internal or external), insufficient fish removal, and poor re-establishment of submerged macrophytes (Van Donk and Gulati, 1995; Søndergaard et al., 2007; Gulati et al., 2008). Repeated manipulations would probably be required in order to avoid create a longer term response (Søndergaard et al., 2008). A further complication for Attenborough Nature Reserve is the inevitable passage of fish from the River Trent and River Erewash during flood events into the connected lakes. Also, a fish-manipulation approach to restoring the connected lakes may be met with considerable opposition because of the popularity of the site with anglers. However, these issues could be overcome to some extent if biomanipulation was restricted to large-scale enclosures.

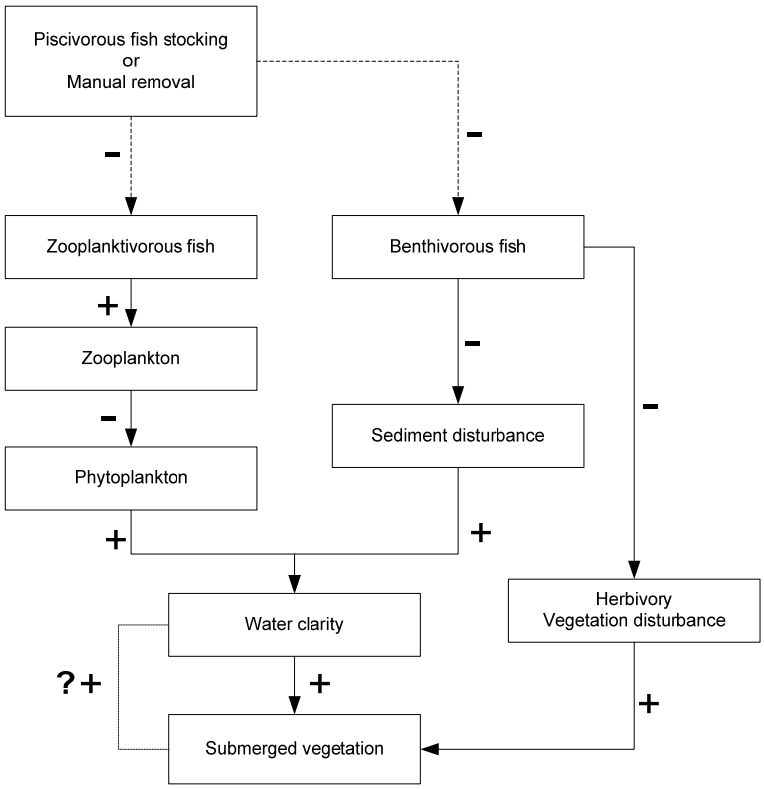

Figure 7.12 The potential cascading effects of biomanipulation on water clarity in shallow lakes. '?' denotes the uncertainty of establishing stable feedback mechanisms to maintain water clarity.

7.5 Prospects for future work

The importance of the River Erewash as a supplier of nutrients to the connected lakes suggests that its diversion will have significant consequences. Analysis of the effects of the River Erewash diversion, which took place during August 2009, is beyond the scope of this thesis. Future studies should use the baseline data presented here to investigate how the diversion affects the water chemistry and plankton community. For example, determining the extent to which reduced N:P ratios and flushing rate may favour cyanobacteria in the short-term is of importance for other shallow lakes. For the wider limnological community, monitoring the effects of the diversion are of high importance because of the continued pressure to reduce nutrient loading to shallow lakes and the potential for anthropogenic interference in the hydrological regime of rivers and lakes associated with them. The likely reduction in N loading, in addition to P loading reduction, that will occur with the diversion of the River Erewash is unusual among studies of nutrient reduction. The data gained by monitoring changes in water chemistry, phytoplankton and zooplankton in Attenborough Nature Reserve after the diversion of the River Erewash would be of significant importance to limnology.

The role of hydrology in determining the structure and function of shallow lakes has considerable potential for future research. The approach taken in this thesis is limited by the division of lakes between those with short WRTs (connected lakes) and those with effectively infinite WRTs (isolated lakes). The effects of intermediate WRTs could be addressed by analysing data from lakes along a continuum of WRTs. Determining how the response of lakes to flooding events varies between seasons, and how the transport of nutrients from point- or diffuse-sources varies during wet periods, would be of benefit for lake management. Extending the temporal span of monitoring the effects of flood events may provide greater insight into the value of flushing for reducing phytoplankton biomass. This thesis has not investigated the longer term response of lakes to flood events, which could be used to develop plans for the manipulations of lake hydrology (Spears et al., 2006). Increasing the quantity of data regarding the responses of lakes to varied WRT and nutrient sources may help to validate the modelling of phytoplankton responses to nutrients and hydrology that is currently being developed (e.g. Jones and Elliot, 2007; Elliott et al., 2009).

Palaeoecological studies may be of interest for determining how lakes respond to the diversion of nutrient rich rivers. This is particularly the case where contemporary monitoring data of high temporal resolution can be supplemented with palaeolimnological datasets to investigate lake responses to nutrient reduction (Battarbee et al., 2005). Combined palaeolimnological and contemporary data has been successfully used in acidification studies, such as the UK Acid Water Monitoring Network (e.g. Monteith, 2005). Such an approach would be particularly powerful at Attenborough Nature Reserve. Palaeolimnological data on the recovery of Lake I2 following its isolation from the River Erewash in 1981, and data gained by monitoring the diversion of the River Erewash from the connected lakes, would provide complementary long- and short-term records of lake responses to nutrient reduction and hydrological change. A multiproxy study would build upon existing palaeoecological data regarding the diversion of the River Erewash into Attenborough Nature Reserve (Sayer and Roberts, 2001). Palaeoecological data would be of considerable importance for informing our understanding of the role of hydrology in affecting shallow lakes, and may further refine an appropriate restoration target (Battarbee et al., 2005).

Estimates of the nutrient budget of the Attenborough Nature Reserve were probably subject to large errors, most likely as a consequence of the low-resolution of measurements at the outflows from the connected lakes. Future monitoring work should ensure that higher resolution measurements of outflows are made in order to estimate more accurately the storage and retention of nutrients. The installation of automated flow-gauges or more frequent monitoring would improve these estimates. Understanding the uptake and release of nutrients may help to more fully quantify the effects of specific flood events on the loss of nutrients from Attenborough Nature Reserve, and to determine the extent to which managing the hydrology of the connected lakes may be of use for lake management. However, major flood events may still exceed the gauging capacity of automatic flow-gauges.

Future monitoring work should include assessments of the fish communities in Attenborough Nature Reserve. Currently, no quantitative evidence exists regarding the biomass or species composition. Fish may play an important role in determining the structure and functioning of shallow lake ecosystems (Lammens, 1999), for example

by determining zooplankton abundance and community composition (e.g. Brooks and Dodson, 1965; Stephen et al., 2004), resuspending sediments and disturbing submerged vegetation (e.g. Körner and Dugdale, 2003; Nurminen et al., 2003). Therefore, the effects of fish may be of importance in determining the response of Attenborough Nature Reserve to the diversion of the River Erewash.

The mesocosm experiment found significant decreases in total phytoplankton biomass in Si enriched treatments, probably associated with grazing. Increases in dinoflagellates were observed in mesocosms without N addition, which be related to higher P concentrations in these mesocosms. Intense grazing by zooplankton on the phytoplankton community occurred, probably because of their protection from zooplanktivorous fish. Further experimental work should more clearly elucidate the effects of N and Si and determine how they may interact to affect phytoplankton communities, particularly since the diversion of the River Erewash will probably significantly affect N and Si supply to the connected lakes. Mesocosm experiments could include zooplanktivorous fish to reduce grazing pressure on the phytoplankton. An alternative to mesocosm experiments would be microcosm experiments (Nandini and Rao, 2000; Camacho et al., 2003; McKee et al., 2003). These could allow for zooplankton to be excluded relatively easily compared to field-based mesocosms. The relative logistical simplicity of microcosm compared to mesocosm experiments may have other advantages. Some studies have suggested that nutrient limitation may vary throughout the year, and microcosm experiments could be repeated seasonally to determine the temporal variability of N and Si limitation. Because it would be easier to increase the number of treatments, the effects of N could be assessed separately as NH_4-N or NO_3-N limitation. Replication also improves the statistical power of experiments (Carpenter, 1996). An improved understanding of nutrient limitation where P is abundant would be useful for the management of both Attenborough Nature Reserve and other eutrophic shallow lakes.

7.6 Conclusions

The lakes of Attenborough Nature Reserve were monitored between March 2005 and March 2008. Water chemistry and plankton biomass and composition datasets have

been used to identify the structure (Chapter 3) and functioning (Chapter 4) of the constituent lakes. Long-term (three years) whole-lake scale monitoring data was complemented with an investigation of the effects of short-term flooding events (one year, Chapter 5). The research also used an experimental approach, which was conducted on smaller spatial and shorter temporal (35 days) scales, to simulate the lake restoration and the effects of N and Si on phytoplankton and zooplankton (Chapter 6). The conclusions of the thesis are:

- **Lakes connected to nutrient-rich inflows in Attenborough Nature Reserve were turbid and devoid of vegetation**, in contrast to the clear-water and abundant macrophyte communities in Lakes I1 and I2.
- **The River Erewash is an important supplier of nutrients to the connected lakes.** High concentrations of P, N and Si were associated with connectivity to the River Erewash. Phytoplankton biomass and minerogenic turbidity were also higher in lakes connected to the River Erewash.
- **Lakes I1 and I2, which were isolated from the River Erewash, had lower concentrations of P, N and Si.** Water clarity was higher, phytoplankton biomass lower and suspended solids concentrations lower in comparison to the connected lakes.
- **The inflow to Lake I3 also supplied nutrient-rich water to the Attenborough Nature Reserve**, and is probably the principal cause of the higher nutrient concentrations and higher phytoplankton biomass in Lake I3 than the other isolated lakes.
- **Internal release of P during the summer was significant in all lakes.** Internal processes reduced in-lake N concentrations in the connected lakes. Si concentrations were strongly mediated by diatoms in the connected lakes but not the isolated lakes.
- **High nutrient loading and short WRTs associated with connectivity to the River Erewash may have favoured small and rapidly reproducing phytoplankton**. Chlorophytes, cryptophytes and small centric diatoms were abundant in the phytoplankton community of the connected lakes.
- **Isolation from the River Erewash appeared to increase proportionally the abundance of cyanobacteria** in lakes I1 and I2 compared to the connected lakes and Lake I3. N-limitation, low N:P ratios and long WRTs probably favoured the growth of cyanobacteria in lakes I1 and I2.

- **In Lake I3, high concentrations of NO_3-N and NH_4-N may have suppressed cyanobacterial growth**. Diatoms and cryptophytes were the most abundant phytoplankton groups in Lake I3.
- **The total abundance of zooplankton was higher in the connected lakes and Lake I3 than in the isolated lakes**. *Daphnia hyalina*, calanoid copepods and *Bosmina* spp. were the most frequently encountered zooplankton taxa. Calanoid copepods were abundant in isolated lakes I1 and I2, and large-bodied *Daphnia* (such as *D. pulex*) were more abundant than in the other lakes.
- **Phytoplankton were probably controlled by a combination of top-down and bottom-up processes in the connected lakes.** Grazing was probably principally responsible for the development of a clear-water period during the spring. During summer, NH_4-N and SiO_3 concentrations were strongly depleted in the connected lakes. P was not limiting for phytoplankton at any time.
- **In isolated lakes I1 and I2, grazing probably exerted a strong control on phytoplankton biomass.** NO_3-N may also have limited phytoplankton growth in lakes I1 and I2. P was probably not limiting to phytoplankton in any of the lakes in Attenborough Nature Reserve.
- **Flooding reduced P concentrations and total phytoplankton biomass** in all the lakes of Attenborough Nature Reserve. **In-lake concentrations of N and Si did not change** during flooding in the connected lakes suggesting that the transport of diffuse-source derived nutrients increased during flooding.
- **Cyanobacterial biomass was lower and diatoms and cryptophytes were the most abundant phytoplankton groups** during flooding in both connected and isolated lakes.
- The mesocosm experiments found that **manipulations of N, Si, and simulated isolation had few significant effects on phytoplankton and zooplankton communities.** Si additions were associated with slightly decreased chlorophyll-*a* concentration. **P concentrations increased** in all mesocosms to which NO_3-N was not added, which may have been associated with a significant increase of mean dinophyte biovolume.
- **The effects of connectivity to rivers may be of benefit to lake management**, by flushing out P, reducing the total abundance of

phytoplankton and reducing cyanobacterial biomass. **Lake flushing may be of benefit to lake restoration.**

- **Lake restoration by isolation is likely to result in short-term increases in water-column P concentration and total phytoplankton biomass.** In the long-term, removing diffuse-source derived N inputs from shallow lakes may be a significant advantage for the use of river diversion for lake restoration.
- **Specific recommendations for future research of value to Attenborough Nature Reserve include continued monitoring of the constituent lakes during river diversion, finer resolution temporal monitoring of nutrient losses from the currently connected lakes and fish community assessments.** Palaeolimnological investigations would be highly complementary to high temporal resolution monitoring, in order to track past responses to river diversion and predict potential recovery trajectories.

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Appendixes

A: Water chemistry

Lake	Date	TP ($\mu g\ L^{-1}$)	SRP ($\mu g\ L^{-1}$)	NH_4-N (mg L-1)	NO_3-N ($mg\ L^{-1}$)	SiO_3 ($mg\ L^{-1}$)	pH	Mg^{2+} ($mg\ L^{-1}$)	Ca^{2+} ($mg\ L^{-1}$)	Na^{+} ($mg\ L^{-1}$)	K^{+} ($mg\ L^{-1}$)
C1	22/03/2005	286	80	0.01	10.97	0.39	9.08				
C1	15/04/2005	415	210	0.00	8.25	0.50	9.21	1.97	3.39	5.17	0.32
C1	12/05/2005	647	554	0.31	8.58	1.86	8.14	2.01	3.67	5.00	0.34
C1	09/06/2005	643	608	0.22	7.06	3.30	8.42	1.37	2.55	4.04	0.24
C1	06/07/2005	990	710	0.07	4.27	5.39	8.59	1.44	2.71	4.36	0.28
C1	01/08/2005	548	452	0.00	3.74	1.07	9.76	1.55	2.76	4.70	0.30
C1	31/08/2005	874	489	0.00	3.33	0.85	9.89	1.49	1.79	4.21	0.28
C1	04/10/2005	767	516	0.00	3.55	0.78	9.47	1.43	2.47	4.06	0.27
C1	01/11/2005	702	495	0.14	6.04	4.69	7.92	2.20	1.58	4.30	0.37
C1	24/11/2005	610	480	0.17	7.30	5.79	7.67	1.05	1.70	2.72	0.21
C1	21/12/2005	610	390	0.31	8.50	5.69	8.48	1.52	1.63	3.16	0.17
C1	17/01/2006	532	430	0.30	9.53	5.22	8.57	1.47	1.97	3.78	0.17
C1	16/02/2006	655	521	0.07	11.49	5.09	8.61	1.43	1.76	3.63	0.22
C1	15/03/2006	443	309	0.03	9.38	3.53	8.6	1.62	2.05	3.88	0.24
C1	20/04/2006	366	188	0.01	9.05	0.82	7.2	1.47	2.76	3.54	0.25
C1	18/05/2006	739	62	0.46	3.76	1.72	8.64	1.37	2.53	3.25	0.23
C1	15/06/2006	547	444	0.17	5.94	3.10	8.52	1.21	2.43	2.59	0.23
C1	13/07/2006	631	291	0.00	3.94	0.84	9.4	1.24	1.86	3.16	0.26
C1	10/08/2006	1158	461	0.00	4.48	3.10	8.63	2.00	0.94	2.24	0.23
C1	07/09/2006	597	388	0.40	5.39	3.57	8.83	1.98	0.96	2.31	0.20
C1	05/10/2006	565	391	0.04	8.32	0.70	9.01	2.05	0.96	2.50	0.19
C1	02/11/2006	511	399	0.24	5.08	5.32	9.14	1.26	1.78	2.15	0.22
C1	30/11/2006	428	344	0.31	5.08	5.89	8.73	1.36	2.73	2.88	0.24
C1	20/12/2006	413	351	0.41	7.28	5.45	8.99	1.31	2.65	2.85	0.23
C1	24/01/2007	274	235	0.34	6.45	6.61	9.2	1.02	2.35	1.95	0.17
C1	15/02/2007	310	218	0.47	6.54	5.46	9.21	1.13	2.44	3.05	0.18
C1	15/03/2007	397	248	0.00	6.16	4.49	8.93	0.97	1.88	3.30	0.14
C1	19/04/2007	317	195	0.25	5.05	0.59	8.55	1.28	1.73	4.50	0.25
C1	17/05/2007	528	371	0.41		3.16	8.15	1.04	1.81	3.92	0.21
C1	14/06/2007	495	296	0.00	4.99	2.33	9.19	1.54	2.61	4.17	0.27
C1	16/07/2007	592	364	0.18	3.24	4.49	7.88	1.22	2.58	2.25	0.21
C1	08/08/2007	389	285	0.00	3.52	3.90	8.09	1.57	2.88	2.57	0.22
C1	11/09/2007	590	261	0.00	4.47	1.15	8.97	1.96	1.92	4.23	0.29
C1	15/10/2007	620	440	0.00	2.99	3.99	8.34	1.99	2.00	4.43	0.29
C1	15/11/2007	554	469	0.00	9.72	4.30	8.07	1.77	2.51	4.75	0.32
C1	13/12/2007	473	356	0.42	7.26	4.53	7.6	1.11	1.34	2.41	0.20
C1	17/01/2008	496	222	0.26	4.23	5.48	9	0.75	1.34	1.33	0.15
C1	19/02/2008	340	236	0.03	7.39	3.89	7.8	0.93	1.46	1.92	0.13
C1	18/03/2008	355	110	0.00	16.51	0.82	9.13	0.94	1.46	2.39	0.12
C2	22/03/2005	192	49	0.01	6.80	0.35	9.3				
C2	15/04/2005	383	181	0.00		0.50	9.38	1.87	3.13	5.26	0.30
C2	12/05/2005	573	500	0.29	6.34	1.52	8.35	2.00	3.57	5.16	0.33
C2	09/06/2005	528	485	0.00	3.90	2.30	8.73	1.61	2.99	4.63	0.27
C2	06/07/2005	906	661	0.01	2.72	4.76	9.03	1.44	2.76	4.41	0.28
C2	01/08/2005	893	514	0.00	0.03	0.81	9.78	1.60	2.73	4.86	0.30
C2	31/08/2005	889	340	0.00	1.16	1.07	10.18	1.52	2.20	4.49	0.28
C2	04/10/2005	822	574	0.00	1.40	0.74	9.54	1.57	2.63	4.72	0.29
C2	01/11/2005	611	459	0.30	6.42	5.24	8.08	2.62	3.93	4.05	0.38
C2	24/11/2005	558	423	0.21	6.43	5.77	7.81	1.25	2.11	2.79	0.22
C2	21/12/2005	434	357	0.30	7.51	5.38	8.33	1.43	1.59	2.89	0.16
C2	17/01/2006	509	416	0.37	7.19	5.13	8.11	1.39	1.87	3.94	0.16
C2	16/02/2006	540	423	0.03	9.35	4.92	8.69	1.44	1.79	3.46	0.22
C2	15/03/2006	442	308	0.15	7.52	3.49	8.95	1.60	2.03	4.01	0.23

Lake	Date	TP ($\mu g\ L^{-1}$)	SRP ($\mu g\ L^{-1}$)	NH_4-N (mg L-1)	NO_3-N ($mg\ L^{-1}$)	SiO_3 ($mg\ L^{-1}$)	pH	Mg^{2+} ($mg\ L^{-1}$)	Ca^{2+} ($mg\ L^{-1}$)	Na^{+} ($mg\ L^{-1}$)	K^{+} ($mg\ L^{-1}$)
C2	20/04/2006	327	151	0.01	5.50	0.65	9.21	1.40	2.66	3.40	0.25
C2	18/05/2006	524	507	0.43	3.49	1.58	8.54	1.43	2.64	3.58	0.28
C2	15/06/2006	462	388	0.31	4.74	3.17	8.38	1.07	2.15	2.20	0.20
C2	13/07/2006	545	214	0.00	2.19	0.88	9.68	1.18	1.65	2.76	0.23
C2	10/08/2006	1219	605	0.09	1.41	3.64	9.65	1.94	0.78	2.18	0.23
C2	07/09/2006	558	301	0.00	3.56	2.57	9.27	1.97	0.99	1.81	0.17
C2	05/10/2006	534	329	0.00	7.29	1.32	8.89	2.28	0.92	2.54	0.19
C2	02/11/2006	421	316	0.29	4.85	5.40	8.59	1.18	1.66	1.96	0.20
C2	30/11/2006	391	287	0.37	4.85	6.12	8.32	1.29	2.69	2.47	0.22
C2	20/12/2006	371	333	0.45	7.01	5.42	8.57	1.32	2.67	2.85	0.23
C2	24/01/2007	233	215	0.36	6.87	6.18	8.62	1.03	2.42	2.19	0.17
C2	15/02/2007	304	178	0.49	5.64	5.17	8.66	1.03	2.32	2.51	0.17
C2	15/03/2007	360	243	0.01	6.40	4.81	8.5	1.05	1.95	2.90	0.17
C2	19/04/2007	255	126	0.08	5.44	0.70	9.1	1.19	1.64	3.83	0.22
C2	17/05/2007	404	311	0.58		3.06	8.46	0.82	1.62	3.15	0.17
C2	14/06/2007	401	248	0.02	4.58	2.23	9.21	1.51	2.76	5.18	0.26
C2	16/07/2007	488	337	0.87	3.25	4.72	7.61	1.18	2.53	2.12	0.20
C2	08/08/2007	429	265	0.02	3.15	4.52	8.03	1.48	2.75	2.35	0.22
C2	11/09/2007	495	235	0.00	2.46	1.79	8.84	1.85	1.87	3.57	0.27
C2	15/10/2007	654	386	0.00	1.49	3.43	8.71	1.99	2.07	3.97	0.27
C2	15/11/2007	474	387	0.00	3.55	3.94	8.32	1.75	2.46	4.18	0.29
C2	13/12/2007	400	267	0.37	6.77	5.09	7.6	1.06	1.35	1.89	0.18
C2	17/01/2008	453	185	0.21	3.69	5.76	8.15	0.67	1.28	0.82	0.15
C2	19/02/2008	341	229	0.14	7.10	4.26	7.82	0.96	1.53	1.96	0.16
C2	18/03/2008	391	182	0.26	12.94	3.18	8.44	0.77	1.33	1.45	0.11
C3	22/03/2005	239	81	0.04	6.33	0.36	9.55				
C3	15/04/2005	350	150	0.00	4.45	0.46	9.38	1.82	3.03	5.38	0.29
C3	12/05/2005	583	450	0.24	5.18	1.52	8.61	1.98	3.39	5.33	0.33
C3	09/06/2005	490	427	0.00	2.99	1.90	9	1.60	2.95	4.58	0.26
C3	06/07/2005	987	724	0.29	1.94	4.28	9.02	1.53	2.75	4.63	0.28
C3	01/08/2005	911	571	0.00	0.51	0.94	9.78	1.66	2.71	5.02	0.30
C3	31/08/2005	814	328	0.00	0.21	1.34	10.42	1.39	1.48	4.25	0.26
C3	04/10/2005	799	611	0.00	0.92	1.64	9.64	1.53	2.50	4.61	0.29
C3	01/11/2005	673	502	0.33	5.98	4.59	8.15	2.39	3.31	4.07	0.36
C3	24/11/2005	500	447	0.24	5.38	5.44	8.1	1.04	1.79	2.34	0.20
C3	21/12/2005	400	353	0.32	6.75	5.24	8.44	1.34	1.55	2.65	0.15
C3	17/01/2006	511	421	0.32	7.28	4.96	8.21	1.40	1.90	3.92	0.18
C3	16/02/2006	468	364	0.03	7.84	4.85	8.66	1.36	1.73	3.43	0.20
C3	15/03/2006	444	312	0.61	8.41	3.48	8.62	1.55	1.95	4.11	0.23
C3	20/04/2006	277	113	0.00	0.74	0.29	8.7	1.28	2.47	3.40	0.22
C3	18/05/2006	479	437	0.47	2.54	1.06	8.51	1.39	2.60	3.54	0.26
C3	15/06/2006	407	329	0.18	4.16	3.34	8.4	1.10	2.26	2.22	0.20
C3	13/07/2006	552	130	0.00	1.03	0.71	9.77	1.13	1.47	2.54	0.22
C3	10/08/2006	1260	545	0.00	0.46	3.74	9.48	1.87	0.70	2.12	0.22
C3	07/09/2006	782	462	0.03	1.39	3.15	9.05	1.95	0.82	2.29	0.21
C3	05/10/2006	580	429	0.02	4.43	1.92	8.89	2.04	0.90	2.52	0.20
C3	02/11/2006	426	316	0.29	4.29	5.00	8.67	1.19	1.65	2.02	0.21
C3	30/11/2006	423	278	0.33	5.08	6.05	8.46	1.25	2.60	2.31	0.22
C3	20/12/2006	428	327	0.45	6.95	5.35	8.63	1.31	2.65	2.84	0.23
C3	24/01/2007	252	219	0.33	6.64	6.27	8.87	1.04	2.40	2.02	0.17
C3	15/02/2007	293	176	0.48	5.60	5.46	8.75	1.06	2.24	2.74	0.19
C3	15/03/2007	329	220	0.00	5.68	4.71	8.55	1.06	2.02	2.41	0.18
C3	19/04/2007	196	88	0.06	3.59	0.54	8.71	1.23	1.74	3.48	0.21

Lake	Date	TP ($\mu g\ L^{-1}$)	SRP ($\mu g\ L^{-1}$)	NH_4-N (mg L-1)	NO_3-N ($mg\ L^{-1}$)	SiO_3 ($mg\ L^{-1}$)	pH	Mg^{2+} ($mg\ L^{-1}$)	Ca^{2+} ($mg\ L^{-1}$)	Na^{+} ($mg\ L^{-1}$)	K^{+} ($mg\ L^{-1}$)
C3	17/05/2007	437	304	0.49		2.94	8.46	0.85	1.62	3.29	0.18
C3	14/06/2007	338	185	0.00	2.08	2.04	9.09	1.48	2.68	3.81	0.25
C3	16/07/2007	518	332	0.80	3.53	4.81	7.31	0.93	2.02	1.62	0.15
C3	08/08/2007	442	236	0.01	2.37	4.56	8.11	1.38	2.69	1.96	0.20
C3	11/09/2007	463	241	0.00	0.95	2.30	8.84	1.73	1.86	2.86	0.24
C3	15/10/2007	556	380	0.00	2.59	2.93	8.59	1.92	2.05	3.50	0.26
C3	15/11/2007	470	355	0.00	1.11	3.86	8.29	1.66	2.43	3.76	0.26
C3	13/12/2007	467	274	0.37	6.40	4.58	7.65	0.96	1.10	1.80	0.17
C3	17/01/2008	468	188	0.21	3.72	5.72	8.6	0.68	1.29	1.00	0.07
C3	19/02/2008	307	218	0.13	6.20	4.07	7.89	0.88	1.43	1.76	0.12
C3	18/03/2008	429	178	0.26	11.67	0.30	8.78	0.79	1.33	1.53	0.11
I1	22/03/2005	95	31	0.01	0.06	0.31	8.75				
I1	15/04/2005	96	41	0.01	0.05	0.36	8.29	0.95	2.19	2.13	0.17
I1	12/05/2005	112	77	0.02	0.06	0.32	8.42	1.04	2.44	2.23	0.18
I1	09/06/2005	96	73	0.00	0.08	0.38	8.75	0.71	1.58	1.59	0.11
I1	06/07/2005	205	118	0.00	0.07	0.91	9.32	0.71	1.17	1.69	0.11
I1	01/08/2005	341	282	0.00	0.00	0.69	9.4	0.89	1.40	2.07	0.14
I1	31/08/2005	390	319	0.00	0.00	0.54	9.14	0.88	0.66	2.01	0.14
I1	04/10/2005	259	283	0.05	0.02	0.47	8.36	0.83	1.28	1.90	0.13
I1	01/11/2005	315	258	0.14	0.13	0.63	8.21	0.93	2.18	2.94	0.22
I1	24/11/2005	280	22	0.12	0.21	0.98	7.54	0.65	1.03	1.73	0.12
I1	21/12/2005	280	142	0.15	0.26	0.68	8.41	0.88	0.98	1.71	0.12
I1	17/01/2006	174	141	0.12	0.00	0.67	8.28	0.78	1.14	1.62	0.09
I1	16/02/2006	166	110	0.06	0.33	0.76	9.07	0.88	1.26	1.76	0.14
I1	15/03/2006	108	74	0.02	0.16	0.40	8.83	0.85	1.26	1.56	0.12
I1	20/04/2006	70	34	0.00	0.00		9.05	0.78	1.78	1.91	0.15
I1	18/05/2006	151	77	0.03	0.00	0.41	8.72	0.77	1.76	1.89	0.14
I1	15/06/2006	80	23	0.00	0.01	1.70	8.97	0.73	1.38	1.84	0.12
I1	13/07/2006	237	106	0.02	0.01	0.40	9.5	0.74	0.83	1.92	0.13
I1	10/08/2006	726	136	0.00	0.00	1.98	9.79	1.02	0.35	1.39	0.12
I1	07/09/2006	481	185	0.10	0.05	3.26	9.58	1.00	0.36	1.40	0.12
I1	05/10/2006	407	209	0.10	0.13	3.50	9.12	1.01	0.38	1.40	0.11
I1	02/11/2006	214	89	0.05	0.20	4.39	8.65	0.75	2.09	1.56	0.14
I1	30/11/2006	168	59	0.02	0.20	4.60	8.47	0.81	1.75	2.05	0.15
I1	20/12/2006	69	80	0.19	0.15	3.98	8.68	0.78	1.78	2.02	0.15
I1	24/01/2007	80	57	0.04	0.57	4.14	8.13	0.69	1.75	1.99	0.13
I1	15/02/2007	89	48	0.00	0.36	3.13	8.76	0.62	1.59	2.48	0.11
I1	15/03/2007	125	23	0.00	0.00	2.39	8.72	0.64	1.30	2.02	0.14
I1	19/04/2007	101	41	0.01	0.00	1.13	9.13	0.57	1.62	2.04	0.13
I1	17/05/2007	93	55	0.04		0.45	8.69	0.51	1.62	2.01	0.13
I1	14/06/2007	98	43	0.00	0.03	0.59	9.21	0.86	2.14	2.19	0.15
I1	16/07/2007	136	83	0.05	0.17	0.72	7.48	0.81	2.02	1.79	0.16
I1	08/08/2007	138	82	0.00	0.02	0.59	8.28	0.92	1.86	1.72	0.15
I1	11/09/2007	204	131	0.00	0.04	1.11	8.83	1.10	1.43	1.81	0.16
I1	15/10/2007	177	115	0.03	1.71	0.53	8.71	1.11	1.46	2.12	0.16
I1	15/11/2007	142	76	0.00	0.00	0.32	8.54	0.96	1.87	1.83	0.15
I1	13/12/2007	164	86	0.16	0.14	0.59	7.73	0.76	1.16	1.64	0.14
I1	17/01/2008	184	106	0.20	0.99	2.02	8.07	0.73	0.46	1.64	0.14
I1	19/02/2008	129	69	0.00	0.85	1.67	8.09	0.61	1.46	1.13	0.09
I1	18/03/2008	105	14	0.00	0.66	0.77	8.44	0.62	1.52	1.16	0.09
I2	22/03/2005	41	1	0.00	0.02	0.25	8.71				
I2	15/04/2005	46	3	0.00	0.00	0.44	8.54	1.32	2.30	2.15	0.13
I2	12/05/2005	59	37	0.01	0.09	0.46	8.73	1.43	2.45	2.31	0.15

Lake	Date	TP ($\mu g\ L^{-1}$)	SRP ($\mu g\ L^{-1}$)	NH_4-N (mg L-1)	NO_3-N ($mg\ L^{-1}$)	SiO_3 ($mg\ L^{-1}$)	pH	Mg^{2+} ($mg\ L^{-1}$)	Ca^{2+} ($mg\ L^{-1}$)	Na^{+} ($mg\ L^{-1}$)	K^{+} ($mg\ L^{-1}$)
I2	09/06/2005	36	5	0.00	0.00	0.57	9.14	1.21	1.55	1.97	0.11
I2	06/07/2005	77	50	0.00	0.04	0.54	8.96	1.21	1.34	2.05	0.11
I2	01/08/2005	116	29	0.00	0.05	0.89	9.23	1.34	1.28	2.24	0.13
I2	31/08/2005	186	48	0.00	0.05	0.86	9.46	1.37	0.57	2.25	0.13
I2	04/10/2005	110	91	0.11	0.04	1.10	8.82	1.33	1.24	2.21	0.13
I2	01/11/2005	102	64	0.12	0.26	1.28	8.27	2.30	0.87		
I2	24/11/2005	77	39	0.07	0.38	1.42	8.05	0.97	1.00	1.89	0.11
I2	21/12/2005	77	4	0.13	0.35	1.61	8.56	1.37	0.95	2.00	0.07
I2	17/01/2006	70	22	0.09	0.32	1.45	8.3	1.27	1.17	2.04	0.08
I2	16/02/2006	47	12	0.05	0.41	1.32	8.95	1.35	1.21	2.00	0.12
I2	15/03/2006	30	0	0.02	0.17	0.31	8.56	1.35	1.27	1.97	0.12
I2	20/04/2006	3	3	0.00	0.60	0.13	8.73	1.22	1.78	2.13	0.13
I2	18/05/2006	85	15	0.02	0.02	0.56	8.65	1.17	1.69	2.14	0.12
I2	15/06/2006	49	14	0.00	0.01	1.32	8.9	1.20	1.09	2.13	0.10
I2	13/07/2006	16	11	0.00	0.01	1.14	9.36	1.17	0.82	2.23	0.10
I2	10/08/2006	199	52	0.11	0.01	0.87	9.29	1.49	0.22	1.69	0.12
I2	07/09/2006	152	61	0.28	0.04	1.33	8.75	0.56	0.91	2.39	0.10
I2	05/10/2006	98	44	0.21	0.48	2.67	8.54	1.61	0.26	1.71	0.09
I2	02/11/2006	63	29	0.06	0.22	2.83	8.48	1.26	1.01	1.91	0.12
I2	30/11/2006	37	18	0.09	0.22	2.28	8.44	1.31	1.74	2.61	0.13
I2	20/12/2006	37	23	0.08	0.37	2.30	8.61	1.28	1.78	2.52	0.12
I2	24/01/2007	36	8	0.07	0.40	2.81	8.56	1.09	1.69	2.45	0.10
I2	15/02/2007	35	8	0.00	0.22	2.18	8.7	1.12	1.75	2.42	0.11
I2	15/03/2007	44	5	0.00	0.06	1.43	8.66	0.97	1.49	2.75	0.10
I2	19/04/2007	38	6	0.00	0.00	1.07	8.82	0.95	1.48	2.45	0.11
I2	17/05/2007	8	2	0.00	0.01	0.55	8.67	0.94	1.48	2.40	0.10
I2	14/06/2007	35	12	0.00	0.07	0.96	8.91	1.33	1.85	2.49	0.12
I2	16/07/2007	115		0.09	0.08	1.60		1.16	1.89	2.03	0.15
I2	08/08/2007	112	28	0.00	0.02	1.33	8.18	1.24	1.70	1.96	0.14
I2	11/09/2007	100	30	0.00	0.03	0.83	8.43	1.52	1.34	2.09	0.15
I2	15/10/2007	98	29	0.07		0.88	8.26	1.51	1.35	2.11	0.15
I2	15/11/2007	88	33	0.06	0.06	1.00	8.2	1.34	1.75	2.15	0.15
I2	13/12/2007	73	14	0.00	0.24	0.96	8.03	1.15	1.11	1.96	0.13
I2	17/01/2008	78	22	0.11	0.45	0.99	8	0.98	1.28	1.77	0.12
I2	19/02/2008	85	6	0.00	0.75	1.69	7.82	0.74	1.14	1.23	0.09
I2	18/03/2008	84	6	0.00		0.04	8.69	0.75	1.15	1.23	0.07
I3	22/03/2005	108	4	0.72	1.20	0.43	8.86				
I3	15/04/2005	145	3	0.00	0.36	1.26	8.5	1.86	2.81	2.46	0.11
I3	12/05/2005	137	79	0.03	0.49	1.29	8.5	1.87	2.93	2.42	0.11
I3	09/06/2005	198	62	0.01	0.12	2.51	8.47	1.30	2.09	1.62	0.07
I3	06/07/2005	295	173	0.18	0.11	3.31	7.79	1.36	2.29	1.73	0.09
I3	01/08/2005	314	210	0.29	0.00	3.54	8.18	1.42	2.33	1.61	0.09
I3	31/08/2005	326	139	0.00	0.00	0.71	9.15	1.26	1.25	1.30	0.09
I3	04/10/2005	226	102	0.08	0.17	3.54	7.41	1.35	2.20	1.48	0.08
I3	01/11/2005	198	80	0.01	0.50	4.24	7.07	2.27	3.37	1.75	0.12
I3	24/11/2005	133	64	0.00	0.51	4.32	8.16	1.13	1.70	1.07	0.08
I3	21/12/2005	133	52	0.13	3.90	4.08	8.26	1.34	1.40	1.04	0.05
I3	17/01/2006	138	91	0.13	1.12	3.68	8.29	1.31	1.70	1.66	0.05
I3	16/02/2006	138	65	0.00	1.18	3.32	8.61	1.49	1.79	1.65	0.09
I3	15/03/2006	86	7	0.01	0.87	1.87	8.8	1.54	1.87	1.92	0.09
I3	20/04/2006	53	8	0.00	1.26	0.23	8.81	1.44	2.03	2.08	0.10
I3	18/05/2006	423	291	0.76	0.18	1.41	8.49	1.37	2.09	1.93	0.10
I3	15/06/2006	335	274	0.17	0.01	2.04	8.59	1.24	1.82	1.67	0.08

Lake	Date	TP ($\mu g\ L^{-1}$)	SRP ($\mu g\ L^{-1}$)	NH_4-N (mg L-1)	NO_3-N ($mg\ L^{-1}$)	SiO_3 ($mg\ L^{-1}$)	pH	Mg^{2+} ($mg\ L^{-1}$)	Ca^{2+} ($mg\ L^{-1}$)	Na^{+} ($mg\ L^{-1}$)	K^{+} ($mg\ L^{-1}$)
I3	13/07/2006	237	206	0.00	0.03	3.14	8.86	1.15	1.44	1.66	0.08
I3	10/08/2006	746	346	0.00	0.06	2.39	8.99	1.59	0.70	1.43	0.04
I3	07/09/2006	285	117	0.01	0.00	0.56	8.69	1.73	0.74	1.01	0.07
I3	05/10/2006	225	132	0.01	0.15	1.15	8.63	1.56	0.63	0.85	0.06
I3	02/11/2006	163	113	0.06	0.45	2.60	8.66	0.99	1.19	0.87	0.07
I3	30/11/2006	174	94	0.08	0.45	2.96	8.62	1.11	2.05	1.17	0.08
I3	20/12/2006	141	111	0.10	1.34	2.98	8.76	0.77	1.56	1.49	0.03
I3	24/01/2007	123	119	0.09	2.99	4.28	8.82	0.94	2.05	1.18	0.06
I3	15/02/2007	299	83	0.00	3.00	3.36	8.81	0.98	2.09	1.97	0.07
I3	15/03/2007	157	64	0.00	2.83	2.01	8.71	0.97	1.86	1.47	0.07
I3	19/04/2007	115	15	0.03	0.32	1.01	8.86	1.04	1.57	1.59	0.08
I3	17/05/2007	68	28	0.01		1.28	9.07	0.84	1.51	1.39	0.07
I3	14/06/2007	197	98	0.04		1.27	8.99	1.34	2.26	1.42	0.08
I3	16/07/2007	297	115	0.03	0.38	4.60		1.05	2.28	0.90	0.11
I3	08/08/2007	124	73	0.02	0.36	0.51	8.51	1.33	2.56	0.90	0.12
I3	11/09/2007	456	238	0.00	0.05	1.48	8.6	1.65	1.70	1.04	0.12
I3	15/10/2007	337	199	0.04	1.03	1.87	8.32	1.70	1.75	1.02	0.12
I3	15/11/2007	232	134	0.08	0.00	2.74	8.18	1.51	2.16	1.06	0.11
I3	13/12/2007	436	126	0.18	2.71	3.40	7.95	1.16	1.10	1.01	0.09
I3	17/01/2008	379	156	0.11	3.96	4.76	8.16	0.89	1.40	1.04	0.09
I3	19/02/2008	181	93	0.02	3.44	3.68	8.15	0.82	1.37	0.84	0.06
I3	18/03/2008	139	12	0.00	1.29	1.13	8.75	0.89	1.41	0.95	0.05

Lake	Date	Cl^- (mg L^{-1})	Total alkalinity (meq L^{-1})	Cond. (mS cm^{-1})	TSS (mg L^{-1})	Secchi depth (cm)	Temp. (°C)	DO (mg L^{-1})	Chl-a (µg L^{-1})
C1	22/03/2005	5.39	3.4	1.16	20	85		31.7	67
C1	15/04/2005	4.02	3.9	0.79	17	52	10.41	24.8	96
C1	12/05/2005	3.47	3.7	1.07	12	134	14.03	8.5	2
C1	09/06/2005	3.78	4.35	1.23	3	185	19.9	8.2	7
C1	06/07/2005	3.38	3.75	0.94	15	60.5	18.13	8.9	66
C1	01/08/2005	2.86	3.85	0.90	61	37	18.64	17.3	287
C1	31/08/2005	2.75	3.65	0.82	32	38	22.19	25.3	234
C1	04/10/2005	2.84	3.85	1.07	25	37	14.09	18.0	215
C1	01/11/2005	2.21	3	0.86	23	49	12.95	10.8	40
C1	24/11/2005	2.33	3.5	0.98	9	93	5.11	10.9	12
C1	21/12/2005	2.93	3.55	1.07	9	69	4.28	9.6	9
C1	17/01/2006	4.42	3.3	0.71	10	80	6.06	11.8	4
C1	16/02/2006	4.01	3.9	1.26	13	68	6.34	9.5	41
C1	15/03/2006	4.04	3.4	1.18	12	75	4.98	11.9	35
C1	20/04/2006	3.96	3.35	1.05	21	56	11.3	10.7	121
C1	18/05/2006	2.98	3.85	1.08	13	96	16.02	9.3	12
C1	15/06/2006	4.01	3.45	0.95	9	130	21.31	11.3	9
C1	13/07/2006	2.45	3.5	0.98	32	36	21.5	16.9	213
C1	10/08/2006	2.62	3.7	0.96	43	27.5	18.36	16.1	218
C1	07/09/2006	2.47	3.4	0.98	25	35	18.87	8.5	175
C1	05/10/2006	3.73	3.15	1.04	36	39	14.59	14.9	107
C1	02/11/2006	3.73	3.95	0.89	8	99	10.26	9.6	12
C1	30/11/2006	2.11	3.1	0.90	11	99	8.54	7.9	7
C1	20/12/2006	2.13	3	0.92	5	155	6.32	11.7	2
C1	24/01/2007	1.63	2.55	0.77	34	28	4.98	13.2	5
C1	15/02/2007	2.88	2.6	0.95	21	43	5.68	8.7	15
C1	15/03/2007	1.91	3.3	0.92	12	67	9.01	10.5	82
C1	19/04/2007	2.97	2.7	1.19	8	155	14.35	9.7	5
C1	17/05/2007	2.49	3.3	1.06	12	113	13.82	4.2	6
C1	14/06/2007	3.15	3.8	1.09	15	54	19.27	11.7	134
C1	16/07/2007	1.60		0.82	23	49	18.82	12.3	94
C1	08/08/2007	1.88		0.87	21	50	18.66	13.1	72
C1	11/09/2007	3.07	4.5	1.09	30	46	17.41	23.6	143
C1	15/10/2007	1.54	3.8	1.12	15	57	14.35	18.5	78
C1	15/11/2007	3.49	4	1.18	8	53	7.23	10.9	36
C1	13/12/2007	2.24	2.8	0.81	5		4.92	11.6	4
C1	17/01/2008	1.21	2.1	0.53	44	25	6.21	12.6	4
C1	19/02/2008	2.10	3.1	0.93	4	70	4.01	16.6	49
C1	18/03/2008	3.05	3.05	0.90	26	50	7.45	25.3	145
C2	22/03/2005	6.26	2.8	1.10	10	100			31
C2	15/04/2005		3.8	1.08	14	58	10.6	26.3	101
C2	12/05/2005	3.81	3.6	1.08	13	156	15.24	8.6	3
C2	09/06/2005	4.09	3.2	1.21	6	118	20.58	13.8	30
C2	06/07/2005	3.57	3.95	0.94	20	47	18.55	12.0	162
C2	01/08/2005	3.73	3.85	0.91	33	37	19.03	18.9	241
C2	31/08/2005	2.94	3.95	0.84	21	35	22.64	28.4	279
C2	04/10/2005	2.93	4.15	1.06	30	38	14.18	15.4	231
C2	01/11/2005	2.00	2.95	0.81	14	67	13.04	10.5	21
C2	24/11/2005	2.06	3.1	0.90	12	77	4.85	11.2	18
C2	21/12/2005	2.74	3.2	0.98	7	104	4.33	11.9	8
C2	17/01/2006	4.71	3.3	0.72	9	90	5.68	13.2	6
C2	16/02/2006	3.95	3.6	1.21	10	70	5.96	10.2	59
C2	15/03/2006	4.19	3.1	1.18	10	80	5.11	12.7	50

Lake	Date	Cl^- (mg L^{-1})	Total alkalinity (meq L^{-1})	Cond. (mS cm^{-1})	TSS (mg L^{-1})	Secchi depth (cm)	Temp. (°C)	DO (mg L^{-1})	Chl-a (µg L^{-1})
C2	20/04/2006	3.17	3.3	1.01	18	55	11.41	11.0	106
C2	18/05/2006	3.20	3.6	1.08	7	120	15.99	9.6	9
C2	15/06/2006	3.95	3.3	0.88	5	195	20.72	7.1	5
C2	13/07/2006	2.23	3.3	0.90	31	30	22.87	18.8	230
C2	10/08/2006	2.59	3.5	0.89	25	37.5	19.43	9.5	190
C2	07/09/2006	2.15	3	0.93	34	27	19.58	14.8	138
C2	05/10/2006	4.13	3.3	1.02	28	37	14.5	9.4	85
C2	02/11/2006	4.13	3.25	0.82	8	73	10.08	9.0	11
C2	30/11/2006	1.79	3	0.84	16	66	8.72	10.6	8
C2	20/12/2006	2.02	3.05	0.92	6	100	6.21	11.2	3
C2	24/01/2007	1.91	2.8	0.85	25	33	4.73	17.5	4
C2	15/02/2007	2.22	2.3	0.85	23	42	6.63	7.9	8
C2	15/03/2007	1.91	3.5	0.93	13	74	9.05	11.2	57
C2	19/04/2007	2.60	3.25	1.07	13	80	14.59	11.9	11
C2	17/05/2007	2.19	3	0.90	6	163	14.56	6.5	13
C2	14/06/2007	3.64	3.8	1.09	15	59	19.25	8.2	99
C2	16/07/2007	1.47		0.83	15	65	18.76	8.4	34
C2	08/08/2007	1.69		0.81	16	60	19.23	10.8	34
C2	11/09/2007	2.60	4.35	0.98	30	42	17.63	20.0	102
C2	15/10/2007	1.41	3.9	1.04	24	55	14.26	13.6	107
C2	15/11/2007	3.12	4.7	1.08	12	58	6.85	13.0	45
C2	13/12/2007	1.73	2.6	0.72	15	57	4.75	1.6	4
C2	17/01/2008	0.91	2.1	0.47	66	18	6.31	12.2	4
C2	19/02/2008	2.40	3.5	0.91	9	85	3.91	15.2	24
C2	18/03/2008	1.94	2.2	0.71	35	36	7.22	19.4	43
C3	22/03/2005	5.06	3	1.09	17	50	10.3	28.1	84
C3	15/04/2005	4.86	3.5	1.07	16	65	10.52	20.7	91
C3	12/05/2005	4.20	3.6	1.08	31	90	15.13	11.8	6
C3	09/06/2005	4.26	4.3	1.18	8	112	20.96	15.3	33
C3	06/07/2005	3.90	4.15	0.98	16	71.4	18.98	10.9	61
C3	01/08/2005	3.32	3.95	0.91	27	34	19.03	18.9	211
C3	31/08/2005	3.26	4	0.87	40	23	23.26	28.9	281
C3	04/10/2005	3.15	4.1	1.04	26	37	14.36	14.0	141
C3	01/11/2005	1.97	2.75	0.79	14	63	13.12	8.9	36
C3	24/11/2005	1.85	2.8	0.83	6	95	5.14	9.5	10
C3	21/12/2005	2.60	3	0.94	7	103	4.38	9.2	8
C3	17/01/2006	4.07	3.3	0.72	11	121	5.3	12.7	7
C3	16/02/2006	3.97	3.5	1.14	14	73	5.61	11.7	66
C3	15/03/2006	4.33	3.2	1.20	12	90	5.16	13.6	59
C3	20/04/2006	3.28	2.25	1.00	20	54	11.16	11.4	123
C3	18/05/2006	3.67	3.55	1.06	8	163	16.38	6.8	7
C3	15/06/2006	3.97	3.2	0.84	7	150	21.16	7.9	6
C3	13/07/2006	2.10	2.45	0.84	38	26	22.5	21.3	286
C3	10/08/2006	2.59	3.05	0.87	30	30	19.06	18.7	250
C3	07/09/2006	2.68	4.7	0.90	31	32	19.98	17.3	202
C3	05/10/2006	3.80	3.15	1.01	25	43	14.84	8.2	98
C3	02/11/2006	3.80	3.2	0.83	10	77	10.25	10.4	13
C3	30/11/2006	2.11	2.9	0.80	22	64	8.49	9.4	9
C3	20/12/2006	2.04	3.05	0.92	7	104	6.43	9.7	3
C3	24/01/2007	1.72	2.7	0.79	34	33	4.63	9.0	4
C3	15/02/2007	2.63	2.9	0.85	23	34	6.64	11.9	8
C3	15/03/2007	1.79	3.3	0.88	14	70	8.83	11.1	40
C3	19/04/2007	2.10	3.1	0.98	11	102	14.78	9.2	10

Lake	Date	Cl^- (mg L^{-1})	Total alkalinity (meq L^{-1})	Cond. (mS cm^{-1})	TSS (mg L^{-1})	Secchi depth (cm)	Temp. (°C)	DO (mg L^{-1})	Chl-a (µg L^{-1})
C3	17/05/2007	2.23	3.1	0.95	8	127	13.9	8.7	15
C3	14/06/2007	2.73	3.8	1.01	20	54	19.13	11.1	85
C3	16/07/2007	1.50		0.76	21	64	17.72	8.8	35
C3	08/08/2007	1.41		0.72	29	61	19.43	11.8	48
C3	11/09/2007	2.11	4.3	0.86	18	49	18.04	18.5	132
C3	15/10/2007	1.14	3.8	0.96	14	58	13.97	12.5	63
C3	15/11/2007	2.81	4.4	1.00	12	55	7.2	12.8	47
C3	13/12/2007	1.65	2.5	0.70	17	36	4.92	10.5	4
C3	17/01/2008	0.92	2.1	0.47	57	20	6.3	12.9	4
C3	19/02/2008	2.32	3.3	0.85	8	91	4.04	15.1	26
C3	18/03/2008	1.97	2.5	0.73	36	32	7.21	19.4	43
I1	22/03/2005	1.77	3.2	0.55	7	182.5		19.6	5
I1	15/04/2005	1.76	2.9	0.55	3	200	10.35	14.0	5
I1	12/05/2005	1.73	2.5	0.55	12	200	14.8	10.5	5
I1	09/06/2005	1.86	2.6	0.59	1	270	19.31	11.9	2
I1	06/07/2005	1.84	1.95	0.44	11	110	18.3	10.6	27
I1	01/08/2005	1.55	1.95	0.44	5	180	17.52	11.6	11
I1	31/08/2005	1.61	1.95	0.43	2	230	20.6	9.6	7
I1	04/10/2005	1.48	2.1	0.52	4	314	13.6	5.9	4
I1	01/11/2005	1.73	2.2	0.51	5	328	12.89	11.3	3
I1	24/11/2005	1.63	2.8	0.54	3	345	5.51	10.5	3
I1	21/12/2005	1.64	2.5	0.54	7	347	4.39	9.8	4
I1	17/01/2006	1.82	2.4	0.34	6	315	5.16	13.5	2
I1	16/02/2006	1.69	2.5	0.56	6	192	5.81	10.8	5
I1	15/03/2006	1.51	2.45	0.57	4	260	4.94	11.5	8
I1	20/04/2006	1.69	2.4	0.77	10	166	11.34	9.1	9
I1	18/05/2006	1.57	2.7	0.58	3	205	16.36	10.5	7
I1	15/06/2006	1.69	2.3	0.55	8	137	20.8	12.6	32
I1	13/07/2006	1.65	1.8	0.50	14	84	22.16	13.0	64
I1	10/08/2006	1.76	1.3	0.51	37	49	18.88	11.9	299
I1	07/09/2006	1.83	2	0.50	5	49	18.5	10.7	28
I1	05/10/2006	2.45	1.9	0.51	34	40	14.39	9.0	65
I1	02/11/2006	1.56	2.6	0.55	10	82	9.82	8.3	46
I1	30/11/2006	1.75	2.45	0.57	8	123	8	10.6	32
I1	20/12/2006	1.43	2.45	0.58	8	235	5.77	10.1	7
I1	24/01/2007	1.76	2.3	0.60	9	228	4.41	12.8	8
I1	15/02/2007	1.69	2.45	0.61	7	127	5.52	8.8	22
I1	15/03/2007	1.67	2.3	0.61	6	117	8.93	7.9	22
I1	19/04/2007	1.80	2.7	0.64	8	138	14.66	6.8	6
I1	17/05/2007	1.72	2.8	0.64	4	305	14.46	7.7	6
I1	14/06/2007	1.86	2.5	1.09	9	205	19.25	8.2	7
I1	16/07/2007	1.49		0.58	2	247	19.25	9.6	3
I1	08/08/2007	1.45		0.54	4	252	19.59	11.9	6
I1	11/09/2007	1.56	2.5	0.55	4	195	18.05	12.2	5
I1	15/10/2007	0.94	3.25	0.55	8	315	14.35	8.7	6
I1	15/11/2007	1.64	2.6	0.56	6	122	7.04	12.8	43
I1	13/12/2007	1.59	2.5	0.56	13	193	4.48	12.1	11
I1	17/01/2008	1.48	2.5	0.55	12	38	5.86	12.5	8
I1	19/02/2008	1.54	2.6	0.55	8	233	3.79	16.3	17
I1	18/03/2008	1.53	2.3	0.54	23	109	7.01	21.0	40
I2	22/03/2005		2.8	0.59	4	107	10.4	17.2	5
I2	15/04/2005	1.62	3.3	0.58	4	160	10.26	15.3	7
I2	12/05/2005	1.65	2.9	0.59	14	155	15	11.0	2

Lake	Date	Cl^- (mg L^{-1})	Total alkalinity (meq L^{-1})	Cond. (mS cm^{-1})	TSS (mg L^{-1})	Secchi depth (cm)	Temp. (°C)	DO (mg L^{-1})	Chl-a (µg L^{-1})
I2	09/06/2005	1.72	2.6	0.59	2	150	19.75	12.5	3
I2	06/07/2005	1.78	2.35	0.49	3	114.5	18.03	10.7	9
I2	01/08/2005	1.72	2.35	0.49	10	90	19.12	11.7	22
I2	31/08/2005	1.73	2.35	0.47	10	68	22.2	12.3	60
I2	04/10/2005	1.67	2.6	0.60	3	88	13.91	9.1	9
I2	01/11/2005	1.87	2.65	0.60	3	208	12.98	10.9	2
I2	24/11/2005	1.76	2.55	0.63	3	155	4.84	11.9	4
I2	21/12/2005	1.77	2.9	0.64	7	180	4.16	10.2	3
I2	17/01/2006	1.77	3.05	0.40	11	165	5.21	13.3	2
I2	16/02/2006	1.67	2.95	0.66	5	158	5.89	10.3	5
I2	15/03/2006	1.68	3.05	0.65	5	128	4.94	11.9	5
I2	20/04/2006	1.66	2.95	0.66	7	162	11.51	8.7	6
I2	18/05/2006	1.59	2.85	0.66	5	195	16.54	11.6	4
I2	15/06/2006	1.67	2.5	0.60	4	192	21.35	9.6	2
I2	13/07/2006		1.85	0.58	22	58	23.36	14.5	80
I2	10/08/2006	1.88	2.6	0.60	11	76	19.11	7.5	32
I2	07/09/2006	2.05	2.65	0.62	29	132	19.38	9.2	13
I2	05/10/2006	2.60	2.45	0.65	8	106	14.46	7.6	7
I2	02/11/2006	1.81	3.6	0.68	6	129	9.62	9.2	5
I2	30/11/2006	1.96	3.05	0.69	3	71	7.97	10.2	5
I2	20/12/2006	1.95	2.9	0.70	3	185	6	13.4	3
I2	24/01/2007	2.00	2.9	0.71	5	145	4.65	9.8	4
I2	15/02/2007	2.20	2.6	0.71	5	124	5.55	7.5	16
I2	15/03/2007	1.78	3	0.71	4	155	8.87	10.5	7
I2	19/04/2007	2.51	2.95	0.73	7	170	14.95	12.3	5
I2	17/05/2007	1.98	3.1	0.72	4	200	14.51	9.6	7
I2	14/06/2007	2.00	2.9	0.70		145	19.19	10.1	4
I2	16/07/2007	1.59			5	140			9
I2	08/08/2007	1.57		0.59	3	115	19.7	11.5	8
I2	11/09/2007	1.72	2.9	0.60	2	91	18.14	12.8	4
I2	15/10/2007	1.37	2.6	0.61	1	95	14.39	9.5	3
I2	15/11/2007	1.82	4	0.62	3		6.66	12.6	5
I2	13/12/2007	1.75	2.8	0.63	8		4.41	18.1	34
I2	17/01/2008	1.68	3.1	0.62	10	91	6.05	12.5	12
I2	19/02/2008	1.70	2.9	0.60	18	74	3.71	16.3	30
I2	18/03/2008	1.61	2.6	0.58	21	40	6.85	22.2	61
I3	22/03/2005	3.22	3.6	0.80		40	10.5	26.5	48
I3	15/04/2005	3.00	3.8	0.74	28	41	10.65	18.3	70
I3	12/05/2005	2.79	3.7	0.73	3	115	14.07	13.9	24
I3	09/06/2005	2.72	3.7	0.77	5	134	19.5	11.6	50
I3	06/07/2005	2.29	3.25	0.58	9	63	18.42	5.1	27
I3	01/08/2005	1.67	3.1	0.54	6	93	17.29	6.2	20
I3	31/08/2005	1.67	2.9	0.46	22	35	19.86	17.3	
I3	04/10/2005	1.24	2.9	0.57	6	195	13.78	7.7	17
I3	01/11/2005	1.16	2.8	0.53	12	105	13.2	11.3	65
I3	24/11/2005	1.04	2.75	0.56	8	125	5.07	11.7	41
I3	21/12/2005	1.33	3.1	0.62	6	177	4.55	9.6	5
I3	17/01/2006	1.83	3.3	0.45	12	80	5.21	11.4	3
I3	16/02/2006	2.20	3.9	0.76	14	79	5.88	9.3	68
I3	15/03/2006	2.52	3.5	0.79	19	100	5.26	12.0	73
I3	20/04/2006	2.45	2.7	0.72	14	55	11.72	9.7	49
I3	18/05/2006	2.05	3.2	0.73	12	147	16.97	7.8	3
I3	15/06/2006	2.20	3.1	0.68	7	154	21.03	4.7	11

Lake	Date	Cl^- (mg L^{-1})	Total alkalinity (meq L^{-1})	Cond. (mS cm^{-1})	TSS (mg L^{-1})	Secchi depth (cm)	Temp. (°C)	DO (mg L^{-1})	Chl-a (µg L^{-1})
I3	13/07/2006	1.92	3.3	0.68	20	66	23.99	13.5	54
I3	10/08/2006	1.70	3.7	0.65	20	55	19.99	10.0	59
I3	07/09/2006	1.70	3	0.61	21	48	19.51	10.9	79
I3	05/10/2006	2.27	2.65	0.56	16	62	14.79	8.5	32
I3	02/11/2006	2.27	3	0.53	2	55	10.45	11.2	85
I3	30/11/2006	1.21	2.8	0.54	13	120	7.97	9.0	9
I3	20/12/2006	1.23	2.7	0.56	4	158	6.02	7.8	3
I3	24/01/2007	1.36	2.65	0.59	18	67	4.66	10.4	5
I3	15/02/2007	2.23	2.7	0.69	21	42	5.85	8.2	66
I3	15/03/2007	1.63	3.15	0.70	18	54	8.94	8.0	25
I3	19/04/2007	1.68	2.9	0.71	14	69	15.31	9.3	20
I3	17/05/2007	1.31	3.1	0.63	6	126	14.99	10.0	35
I3	14/06/2007	1.64	3.05	0.64	7	60	19.08	6.2	29
I3	16/07/2007	0.92			14	60			69
I3	08/08/2007	0.95		0.54	20	60	19.65	14.6	85
I3	11/09/2007	1.15	3.7	0.57	21	49	18.3	13.9	64
I3	15/10/2007	0.56	3.6	0.57	6	92	14.21	7.6	24
I3	15/11/2007	0.77	3.8	0.59	3	205	6.95	13.5	24
I3	13/12/2007	1.28	3.1	0.58	7		4.7	15.5	7
I3	17/01/2008	1.37	2.7	0.55	67	16	6.05	11.2	8
I3	19/02/2008	1.57	3.3	0.63	12	60	3.86	16.6	45
I3	18/03/2008	1.72	3.1	0.63	23		7.32	21.7	78

B: Phytoplankton species

	Form* / Biovolume		Lake					
Bacillariophyceae			**C1**	**C2**	**C3**	**I1**	**I2**	**I3**
Asterionella formosa Hass. 1850	C	240	X	X	X	X	X	X
Aulacoseira spp. (filament)	F	5625	X	X	X		X	X
Large centric diatom (Stephanodiscus / Cyclotella types)	C	1800	X	X	X	X	X	X
Small centric diatom (Stephanodiscus / Cyclotella types)	C	397	X	X	X	X	X	X
Cocconeis sp.	C	1560	X	X	X	X	X	X
Cymbella sp.	C	1980				X		X
Diatoma sp.	C	1387				X		X
Fragilaria sp.	C	205		X	X	X	X	X
Gyrosigma sp.	C	2006	X	X	X	X	X	X
Meridion sp.	C	1580	X	X	X	X	X	X
Navicula sp.	C	5023	X	X	X	X	X	X
Nitzschia sp.	C	1450	X	X	X	X	X	X
Small un-identified pennate species	C	297	X	X		X	X	X
Synedra spp.	C	110	X	X	X	X	X	X
Tabellaria sp.	C	720	X	X	X	X	X	X
Chlorophyceae								
Actinastrum hantzschii Lagerheim	Cn	713	X	X	X	X	X	
Ankistrodesmus falcatus (Corda) Ralfs	Cn	200	X	X	X	X	X	
Ankyra ancora (G. M. Smith) Fott	C	723	X		X	X	X	
Ankyra judyaii (G. M. Smith) Fott	C	679	X	X	X	X	X	X
Chlamydomonas spp.	C	1410	X	X	X	X	X	X
Chlorella vulgaris Beijerinck	C	1376	X		X	X	X	X
Chlorococcum sp.	C	500			X	X		
Closterium acutum var. *variable* (Lemmerman) Willi Krieger	C	500			X			
Closterium sp.	C	3023	X	X	X	X	X	X
Coelastrum microporum Nägeli in A. Braun	Cn	2974	X	X	X	X	X	X
Cosmarium sp.	C	33513	X			X		
Crucigeniella rectangularis (Nägeli) Komárek	Cn	483	X	X		X		X
Dictyosphaerium pulchellum H. C. Wood	Cl	1103	X	X	X	X	X	X
Eudorina elegans Ehrenburg	Cn	2803	X	X	X	X	X	
Lagerheimia sp.	C	472				X	X	
Micratinium pusillum Fresenius	C	1893	X	X		X	X	X
Monoraphidium contortum (Thuret) Komárková-Legnerova	C	45	X	X	X	X	X	X
Pediastrum boryanum var. *longicorne* Reinsch	Cn	2821	X	X		X		X
Pediastrum duplex Meyen	Cn	2813	X		X	X	X	X
Pediastrum simplex Meyen	Cn	2886	X		X		X	X
Pediastrum sp.	Cn	2764	X	X	X	X	X	X
Pteromonas sp.	C	164	X	X	X	X		
Quadrigula sp.	Cn	523				X		X
Scenedesmus abundans (Kirchner) Chodat	Cn	489**			X			
Scenedesmus arcuatus (Lemmerman) Lemmerman	Cn	461**		X			X	
Scenedesmus bicaudatus Dedusenko	Cn	510**					X	
Scenedesmus communis E. H. Hegewald	Cn	312**	X	X	X	X	X	X
Scenedesmus falcatus Chodat	Cn	607**	X	X	X	X	X	X
Scenedesmus opoliensis P. G. Richter	Cn	770**	X	X	X	X	X	X
Schroderia robusta Korshikov	C	754	X			X	X	X
Selanastrum sp.	Cn	380	X	X	X	X	X	X
Staurastrum planctonicum Teiling	C	1521	X	X	X			
Tetradesmus sp.	Cn	502		X	X			X
Tetraedron caudatum (Corda) Hansgirg	C	487	X	X	X		X	X

Tetraedron incus (Teiling) G. M. Smith	C	500					X	
Tetraedron minimum (A. Braun) Hansgirg	C	500						X
Tetraedron regulare Kützing	C	487		X	X		X	
Tetraedron triangulare Korschikoff	C	500	X	X	X	X	X	X
Tetrastrum elegans Playfair	Cn	390	X	X	X	X		X
Ulothrix sp.	F	430	X			X	X	X
Un-identified green flagellates	C	65	X	X	X	X	X	X
Cyanobacteria								
Anabaena aequalis (Kützing) Bornet et Flahault	T	3100		X				X
Anabaena flos-aquae (Lyngbye) Brébisson Bornet et Flahault	T	891	X		X	X	X	X
Aphanizomenon flos-aquae (Linnaeus) Ralfs ex Bornet et Flahault	T	1630	X		X	X	X	
Aphanocapsa sp.	C	2000	X				X	
Chroococcus sp.	C	500				X	X	
Merismopedia sp.	C	64	X	X	X	X	X	X
Microcystis sp.	C	14000	X	X	X	X	X	X
Oscillatoria arghardii Gormont	T	380	X	X	X	X	X	X
Oscillatoria limnetica Lemmerman 1900	T	250	X	X	X	X	X	
Oscillatoria redekei Goor 1918	T	308			X			
Oscillatoria sp.	T	258	X	X	X	X		X
Spirulina sp.	T	6250						X
Cryptophyceae								
Cryptomonas spp.	C	1149	X	X	X	X	X	X
Rhodmonas spp.	C	361	X	X	X	X	X	X
Euglenophyceae								
Euglena sp.	C	500	X	X	X	X	X	X
Euglena acus Ehrenburg 1830	C	700			X			
Phacus sp.	C	558	X	X	X	X	X	X
Phacus caudatum	C	221				X		X
Trachelomonas sp.	L	980	X	X	X	X	X	X
Trachelomonas bacillifera Playfair 1915	L	980					X	X
Dinophyceae								
Ceratium hirundinella (O. F. Müller) Dujardin 1814	C	6500					X	
Gymnodinium sp.	C	978	X	X	X	X	X	X
Peridinium sp.	C	6250	X	X	X	X	X	X
Chrysophyceae								
Dinobryon cylindricum O. E. Imhof 1887	L	1800				X	X	
Mallomonas sp.	C	1491	X	X	X	X	X	X

***: Form abbreviations:** C, cell; Cl, colony; Cn, coenobial; F, filament; L, lorica; T, trichome.

**: Biovolume estimate for two-celled coenobia.

C: Phytoplankton dataset

(all data presented as $\mu m^3\ mL^{-1}$)

Lake	Date	Asterionella sp.	Aulacoseira individuals	Small centric diatoms	Large centric diatoms	Synedra sp.	Cymbella sp.	Diatoma sp.	Fragilaria sp.	Gyrosigma sp.	Meridion sp.	Nitzschia sp.	Cocconeis	Navicula sp.	Small pennate diatoms	Tabellaria	Green Flagellate	Actinastrum hantzschii
C1	22/03/2005	0	0	623012	150585	2347	0	0	0	0	0	0	0	0	0	0	0	0
C1	15/04/2005	0	0	784061	306888	20246	0	0	0	0	0	0	9571	0	0	0	0	0
C1	11/05/2005	0	0	216363	5592	0	0	0	0	0	1484	0	0	4717	0	0	0	0
C1	09/06/2005	0	0	0	340	0	0	0	0	0	0	0	0	0	0	0	386	0
C1	06/07/2005	0	0	0	8575	103	0	0	0	0	0	0	14161	4717	0	0	0	0
C1	01/08/2005	0	1730906	0	6173789	0	0	0	0	0	0	784317	0	905660	0	0	0	32139
C1	31/08/2005	0	973634	0	4133754	74375	0	0	0	0	0	0	0	0	0	0	0	0
C1	04/10/2005	0	1907529	0	4838234	43714	0	0	0	0	0	480194	0	0	0	0	0	0
C1	01/11/2005	0	506290	6491	336427	2380	0	0	0	0	0	0	5625	0	0	0	0	0
C1	24/11/2005	0	0	0	1670	17	0	0	0	0	0	0	234	1509	0	0	0	0
C1	21/12/2005	0	0	2445	2506	53	0	0	0	0	0	0	125	1204	47	0	99	0
C1	17/01/2006	131	56	423	4567	69	0	0	0	0	0	0	0	0	0	0	173	0
C1	15/02/2006	0	10303	0	335177	2519	0	0	0	0	0	0	0	14376	1700	0	186	0
C1	15/03/2006	1264	0	0	89659	10877	0	0	0	0	0	0	0	0	0	0	152	0
C1	20/04/2006	10729	64373	0	780874	42781	0	0	0	0	0	0	0	0	0	0	3487	0
C1	18/05/2006	0	0	211	792	26	0	0	0	0	0	0	0	1179	0	0	0	0
C1	15/06/2006	0	0	0	761	3	0	0	0	0	0	104	0	0	0	0	0	0
C1	13/07/2006	0	0	0	1362183	743	0	0	0	0	0	0	0	0	2006	0	0	0
C1	10/08/2006	0	0	0	731708	2204	0	0	0	0	0	0	0	704402	0	0	0	0
C1	07/09/2006	0	0	0	3880	10	0	0	0	0	0	0	0	0	0	63	6	0
C1	04/10/2006	0	2705	0	546917	0	0	0	0	0	0	0	0	0	837	125761	1282	0
C1	02/11/2006	38	845	0	4598	34	0	0	0	0	0	0	244	0	0	225	0	0
C1	30/11/2006	0	0	0	410	2	0	0	0	0	0	0	0	0	14	23	8	0
C1	20/12/2006	6	103	0	127	1	0	0	0	0	0	0	0	0	1	0	6	0
C1	24/01/2007	11	0	0	993	0	0	0	0	45	0	0	35	226	0	16	35	0
C1	15/02/2007	0	0	106	268	6	0	0	0	0	0	0	0	1621	0	0	6	0
C1	15/03/2007	0	54091	1744428	498079	826	0	0	0	0	0	0	0	0	2231	0	8301	0
C1	19/04/2007	0	106	270	887	6	0	0	0	0	0	0	0	472	0	0	13	0
C1	17/05/2007	169	0	10565	6276	0	0	0	0	0	0	0	0	0	0	0	20	0
C1	14/06/2007	0	373876	0	1494594	8925	0	0	0	0	0	0	0	81509	0	46734	20041	11570
C1	16/07/2007	0	282155	0	1120814	7286	0	0	0	0	0	0	0	0	0	0	4843	0
C1	08/08/2007	71400	117972	233672	234067	0	0	0	0	0	0	0	0	0	0	0	7735	0
C1	11/09/2007	0	1432137	6616740	1669717	14571	0	0	0	0	0	0	0	0	0	95376	23679	0
C1	15/10/2007	0	15847	0	15658	0	0	0	0	0	0	0	0	28302	0	0	2564	0
C1	15/11/2007	0	15847	0	8575	930	0	0	0	0	0	0	0	14151	0	0	1099	0
C1	13/12/2007																	
C1	17/01/2008	5	0	139	406	2	0	0	0	0	0	0	0	276	0	8	12	0
C1	19/02/2008	613029	0	148750	417551	0	0	0	0	0	0	0	0	0	0	0	0	0
C1	18/03/2008	0	760652	0	4777975	109083	0	0	0	0	0	0	0	0	13387	32454	26369	32139
C2	22/03/2005	795	0	0	86773	3096	0	0	0	0	0	0	0	0	0	0	4843	0
C2	15/04/2005	539	0	141595	26768	13598	0	0	0	0	0	48884	0	11289	0	0	0	0
C2	11/05/2005	0	0	4720	112	21	0	0	0	0	0	407	9496	470	0	0	0	0
C2	09/06/2005	0	0	0	7307	0	0	0	0	0	0	0	9571	15408	0	0	73575	0
C2	06/07/2005	0	978503	299393	49927	1339	0	0	22454	0	0	0	44300	20377	0	0	0	0
C2	01/08/2005	0	8110938	11587054	2896334	47207	0	0	0	0	0	622268	0	0	0	0	0	0
C2	31/08/2005	0	0	2424350	528262	0	0	0	0	0	0	0	0	0	0	23367	0	0
C2	04/10/2005	0	0	22069046	0	79333	0	0	0	0	0	0	0	0	0	0	23439	42852
C2	01/11/2005	0	0	0	58295	0	0	0	0	0	0	0	0	0	890	0	0	0
C2	24/11/2005	0	2761	0	4567	42	0	0	0	0	0	0	0	0	0	0	0	0
C2	21/12/2005	0	8984	8310	594	27	0	0	0	0	0	0	0	2507	0	0	0	0
C2	17/01/2006	0	304	0	2583	37	0	0	35	0	0	0	395	0	201	0	0	0
C2	15/02/2006	506	0	0	20700	261	0	0	0	0	832	0	0	5291	0	0	0	0
C2	15/03/2006	0	0	0	12582	7128	0	0	1011	0	0	0	0	0	0	0	0	0
C2	20/04/2006	0	675911	0	365591	90479	0	0	2749	0	0	0	0	0	2655	19312	0	0
C2	18/05/2006	0	0	307	169	19	0	0	0	0	0	0	0	3852	0	0	0	0
C2	15/06/2006	0	0	794	136	0	0	0	0	0	232	0	0	246	116	0	0	0
C2	13/07/2006	0	0	0	4970797	4225	0	0	0	0	0	0	0	578765	0	0	0	54769
C2	10/08/2006	0	0	1051525	180382	7140	0	0	0	0	0	0	0	4890565	0	0	0	0
C2	07/09/2006	0	0	0	2518906	21420	0	0	0	0	0	23529	0	407547	0	0	0	0
C2	04/10/2006	0	26964	0	944858	2380	0	0	0	0	0	0	0	0	0	0	0	0
C2	02/11/2006	0	3017	0	6212	31	0	0	0	0	0	0	218	1403	207	0	0	0
C2	30/11/2006	0	293	0	166	17	0	0	0	0	247	0	0	786	0	0	0	0
C2	20/12/2006	0	88	0	692	3	0	0	0	0	39	0	38	247	0	0	0	0
C2	24/01/2007	0	136	24	110	0	0	0	0	13	10	0	20	632	0	0	0	0
C2	15/02/2007	0	205	341	5615	12	0	0	0	0	0	0	0	0	67	0	0	0
C2	15/03/2007	0	0	0	82981	494	0	0	0	0	0	0	0	16934	0	62302	0	0
C2	19/04/2007	0	778	1014	4753	31	0	0	0	0	0	0	0	0	0	304	549	0
C2	17/05/2007	144	4512	16604	9235	44	0	0	0	0	0	582	0	9066	0	0	13	0
C2	14/06/2007	0	2831488	393428	552190	0	0	0	0	0	0	0	0	0	0	0	2153	0
C2	16/07/2007	0	544694	43273	122163	264	0	0	0	0	0	0	0	24151	0	0	469	0
C2	08/08/2007	0	28524	0	70864	223	0	0	0	0	0	0	0	0	0	0	5406	0
C2	11/09/2007	0	0	4290076	1842071	2789	0	0	0	0	0	0	0	509434	0	182556	14833	0
C2	15/10/2007	0	0	0	23487	620	0	0	0	0	0	0	0	0	0	0	2381	0

Lake	Date	Asterionella sp.	Aulacoseira individuals	Small centric diatoms	Large centric diatoms	Synedra sp.	Cymbella sp.	Diatoma sp.	Fragilaria sp.	Gyrosigma sp.	Meridion sp.	Nitzschia sp.	Cocconeis	Navicula sp.	Small pennate diatoms	Tabellaria	Green Flagellate	Actinastrum hantzschii
C2	15/11/2007	0	2065	1268	3355	232	0	0	0	0	0	0	0	3538	0	1775	504	0
C2	13/12/2007	0	0	277	804	31	0	0	32	44	278	0	103	0	52	0	27	31
C2	17/01/2008	4	157	0	186	1	0	0	0	0	385	0	0	480	21	0	5	0
C2	19/02/2008	70769	10565	37187	65615	1859	0	0	0	0	0	0	0	0	558	1352	0	0
C2	18/03/2008	9520	0	25964	495335	7140	0	0	0	0	0	0	0	0	1071	5193	2578	0
C3	22/03/2005	0	0	28613	66789	0	0	0	0	0	0	0	0	79846	0	0	0	0
C3	15/04/2005	0	0	48631	104577	0	0	0	0	0	0	68556	0	67854	0	0	0	0
C3	11/05/2005	0	0	12735	85	0	0	0	0	0	0	0	21070	6461	0	0	0	0
C3	09/06/2005	0	0	0	10308	0	0	0	0	0	0	0	0	0	0	0	422752	0
C3	06/07/2005	0	0	12518	263	0	0	0	0	0	0	0	0	1663	0	0	0	0
C3	01/08/2005	0	1419883	0	1373444	22312	0	0	0	0	0	0	87898	56604	0	0	0	8035
C3	31/08/2005	0	0	20029050	2839833	0	0	0	0	0	0	0	0	0	0	0	0	0
C3	04/10/2005	0	0	1828048	327224	0	0	0	0	0	0	682942	0	147863	0	0	0	0
C3	01/11/2005	0	0	86545	23224	0	0	0	0	0	0	2324	0	8050	0	0	0	571
C3	24/11/2005	0	0	20484	1129	0	0	0	0	0	0	688	0	0	0	0	0	0
C3	21/12/2005	0	0	0	1960	61	0	0	0	0	0	0	0	936	0	0	0	0
C3	17/01/2006	0	2536	1268	10485	194	0	0	0	0	0	0	0	7075	0	0	69	251
C3	15/02/2006	5677	17032	0	445133	9888	0	0	0	0	0	0	0	23764	2810	0	8918	0
C3	15/03/2006	225	0	0	52194	18284	0	0	0	0	0	0	0	4717	0	0	2625	0
C3	20/04/2006	17883	0	0	601492	350624	0	0	0	0	0	0	0	83173	0	11922	13454	0
C3	18/05/2006	0	0	46	153	0	0	0	0	0	41	0	0	258	0	0	0	0
C3	15/06/2006	0	0	0	124	0	0	0	21	209	0	0	0	0	0	0	0	0
C3	13/07/2006	0	0	298051	3326286	0	0	0	0	0	0	0	0	0	0	0	10763	0
C3	10/08/2006	0	0	0	44583	0	0	0	0	0	0	0	0	451263	0	0	8759	32028
C3	07/09/2006	0	0	0	2668917	40071	0	0	0	0	0	0	0	1829803	413099	0	21526	0
C3	04/10/2006	0	0	222352	581805	4941	0	0	0	0	0	0	26278	197428	0	32342	1460	0
C3	02/11/2006	0	42198	0	2556	0	0	0	0	0	0	0	1884	6065	0	0	209	0
C3	30/11/2006	296	634	158	1713	48	0	0	0	0	0	0	0	1769	0	127	143	0
C3	20/12/2006	0	317	238	664	2	0	0	0	0	35	0	34	332	0	0	19	0
C3	24/01/2007	34	464	155	392	3	0	0	0	0	23	0	67	216	68	0	17	0
C3	15/02/2007	0	0	211	12582	0	0	0	24	0	0	0	0	2358	174	0	46	0
C3	15/03/2007	25387	0	134145	76352	397	0	0	0	0	0	0	0	0	357	0	156	0
C3	19/04/2007	0	0	14579	4614	77	0	0	0	0	0	0	0	0	209	0	114	0
C3	17/05/2007	423	0	8240	8948	155	0	0	0	0	0	0	0	0	418	0	252	0
C3	14/06/2007	177418	0	0	787382	992	0	0	0	0	14244	0	0	0	0	0	2930	0
C3	16/07/2007	67505	0	6491	147455	0	0	0	0	0	0	0	0	0	0	865	2110	0
C3	08/08/2007	0	0	0	22120	0	0	0	0	0	0	0	0	0	93	0	732	0
C3	11/09/2007	4043	0	0	548368	618	0	0	0	0	0	0	0	28204	0	36385	1460	0
C3	15/10/2007	0	0	0	15099	0	0	0	0	0	0	0	0	0	0	0	1373	1004
C3	15/11/2007	0	0	29805	52590	0	0	0	0	0	0	0	0	13862	0	0	1794	0
C3	13/12/2007	65	162	0	412	77	0	0	0	0	0	0	0	113	0	0	78	0
C3	17/01/2008	0	0	0	1253	21	0	0	0	0	475	0	59	377	0	0	85	0
C3	19/02/2008	403878	0	40568	217723	69417	0	0	0	0	0	0	0	0	0	0	1465	0
C3	18/03/2008	18256	0	0	1288443	64148	0	0	0	0	0	0	0	0	15061	36511	232379	0
I1	22/03/2005	0	0	0	19876	0	0	0	0	0	0	0	0	1175	347	0	0	0
I1	15/04/2005	0	0	49	457	6	0	0	0	0	0	40	0	275	0	0	0	0
I1	11/05/2005	0	0	738	136	8	0	0	0	0	0	297	639	0	20	0	0	0
I1	09/06/2005	0	360	0	178	0	0	0	0	0	0	0	0	125	0	0	0	0
I1	06/07/2005	0	0	0	885	0	0	0	0	0	0	0	0	0	139	0	0	0
I1	01/08/2005	0	0	423	3775	39	0	0	0	0	0	0	0	1179	0	0	92	84
I1	31/08/2005	0	0	453	799	8	0	0	0	0	40	0	0	0	22	0	7	90
I1	04/10/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I1	01/11/2005	0	0	0	256	0	0	0	0	0	0	0	0	173	0	0	0	0
I1	24/11/2005	0	0	0	197	0	0	0	0	0	0	0	0	0	13	0	0	0
I1	21/12/2005	0	0	0	30	0	0	0	10	0	0	0	0	505	22	0	0	0
I1	17/01/2006	0	0	0	25	1	0	0	2	16	12	23	24	118	37	0	0	0
I1	15/02/2006	37	0	0	670	0	0	0	94	308	0	222	0	0	46	0	0	0
I1	15/03/2006	276	0	0	2131	0	0	0	0	0	2423	0	0	0	911	0	50	0
I1	20/04/2006	0	1521	0	447	0	0	0	462	0	0	817	0	0	167	0	0	0
I1	18/05/2006	0	0	0	671	0	0	0	347	0	890	0	0	0	167	0	0	0
I1	15/06/2006	0	0	0	2195	0	0	0	43	0	0	0	0	0	341	0	0	0
I1	13/07/2006	0	0	0	200128	1056	0	0	0	0	0	0	0	0	6416	0	0	10269
I1	10/08/2006	0	0	0	1678	0	0	0	0	0	0	0	0	28302	0	0	0	2009
I1	07/09/2006	0	0	0	2796	0	0	0	0	0	0	0	0	7075	2092	0	0	0
I1	04/10/2006	0	0	20918	699	697	0	0	0	0	0	0	0	0	628	0	0	0
I1	02/11/2006	0	0	0	0	386	0	0	0	0	0	0	1095	10576	417	0	0	0
I1	30/11/2006	0	0	0	146	10	0	0	0	0	0	0	0	0	27	0	0	0
I1	20/12/2006	0	43	0	76	0	71	0	0	0	19	69	149	60	11	0	0	0
I1	24/01/2007	0	453	0	150	6	0	0	0	0	0	0	0	0	75	0	0	0
I1	15/02/2007																	
I1	15/03/2007	353	15896	0	7304	0	0	0	0	0	0	0	0	3697	0	0	1052	0
I1	19/04/2007	0	0	0	594	0	0	0	0	0	0	0	69	0	65	0	9	0
I1	17/05/2007	0	0	136	1218	4	0	0	0	0	0	0	118	951	0	0	39	0

Lake	Date	Asterionella sp.	Aulacoseira individuals	Small centric diatoms	Large centric diatoms	Synedra sp.	Cymbella sp.	Diatoma sp.	Fragilaria sp.	Gyrosigma sp.	Meridion sp.	Nitzschia sp.	Cocconeis	Navicula sp.	Small pennate diatoms	Tabellaria	Green Flagellate	Actinastrum hantzschii
I1	14/06/2007	0	0	158	3530	10	0	0	0	0	696	0	0	1769	0	63	109	0
I1	16/07/2007	14	0	0	268	0	0	41	0	0	93	0	0	0	52	0	15	0
I1	08/08/2007	0	1826	3651	12280	0	0	0	0	0	0	0	0	2547	0	0	264	0
I1	11/09/2007	0	0	2536	4893	0	0	0	0	0	0	0	0	8844	0	0	229	0
I1	15/10/2007	0	0	217	96	0	0	0	0	0	1048	0	0	0	0	0	24	0
I1	15/11/2007	0	0	0	1258	0	0	0	0	0	40061	0	0	0	0	0	1854	0
I1	13/12/2007	0	0	634	6711	39	0	0	0	0	556	0	0	12382	0	0	664	0
I1	17/01/2008	0	317	0	350	0	0	0	0	0	835	0	0	1327	52	0	46	0
I1	19/02/2008	951	0	4279	58194	0	0	0	0	0	0	0	0	0	0	0	1082	0
I1	18/03/2008	12396	0	0	82019	103	0	0	0	0	0	0	0	0	279	0	1404	0
I2	22/03/2005	0	0	1570	1500	16	0	0	0	0	0	3372	0	0	216	0	0	0
I2	15/04/2005	535	0	0	362	0	0	0	0	0	0	0	474	0	90	0	0	0
I2	11/05/2005	0	0	0	91	0	0	0	0	0	0	133	430	2076	0	0	0	0
I2	09/06/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	2240	0	0	0
I2	06/07/2005	0	0	386	341	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	01/08/2005	0	0	634	315	0	0	0	18	0	0	0	0	2211	0	0	109	0
I2	31/08/2005	0	0	5071	31037	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	04/10/2005	0	0	0	124	0	0	0	0	0	330	0	0	0	0	0	10	0
I2	01/11/2005	0	0	0	34	0	0	0	0	0	334	0	132	0	13	0	17	0
I2	24/11/2005	0	0	0	245	0	0	0	0	0	696	0	0	3096	0	63	11	0
I2	21/12/2005	0	0	0	315	0	0	0	0	0	139	0	0	1327	0	0	23	0
I2	17/01/2006	42	0	0	105	0	0	0	0	0	556	0	0	1327	0	0	86	0
I2	15/02/2006	0	0	0	2656	39	0	0	0	0	0	0	549	0	732	0	0	0
I2	15/03/2006	0	0	0	17615	0	0	0	0	0	0	0	0	1769	209	0	0	0
I2	20/04/2006	0	0	0	3579	0	0	0	0	1130	1780	0	0	0	335	0	0	0
I2	18/05/2006	0	0	0	15339	1417	0	0	0	0	0	0	0	0	1912	0	132683	0
I2	15/06/2006	0	0	0	27	0	0	0	0	0	36	67	0	0	109	0	3	0
I2	13/07/2006	0	8282	0	21007	422	0	0	0	0	1212	0	0	0	228	0	199	0
I2	10/08/2006	0	0	0	5369	310	0	0	0	0	0	0	0	0	837	0	293	803
I2	07/09/2006	0	0	32	97	2	0	0	0	0	0	0	0	0	21	0	0	0
I2	04/10/2006	0	308	0	100	3	0	0	0	0	0	0	549	1390	0	0	0	0
I2	02/11/2006	0	0	0	4666	0	0	0	0	0	0	0	0	0	48	0	0	0
I2	30/11/2006	0	0	4327	1193	0	0	0	0	0	0	0	0	2013	59	0	0	0
I2	20/12/2006	0	0	481	93	0	0	0	0	0	0	0	0	0	89	0	2	0
I2	24/01/2007	0	0	0	166	0	0	0	0	0	0	0	0	0	124	0	16	0
I2	15/02/2007																	
I2	15/03/2007	545	282	282	186	0	0	0	0	0	0	0	0	393	93	0	0	0
I2	19/04/2007	19	0	3240	0	9	0	0	0	0	0	227	0	393	232	0	0	0
I2	17/05/2007	0	0	0	20380	103	0	0	0	0	495	0	0	0	93	0	1038	0
I2	14/06/2007	0	0	0	400	0	0	0	0	0	55	0	0	1048	0	0	32	0
I2	16/07/2007	0	0	0	30074	0	0	0	0	0	495	0	0	3145	0	0	570	0
I2	08/08/2007	0	3522	6198	8948	69	0	0	0	0	1978	0	0	6289	0	0	0	0
I2	11/09/2007	0	0	282	62	0	0	0	0	0	1484	0	0	0	0	0	5	0
I2	15/10/2007	0	0	0	124	0	0	0	0	0	124	0	305	197	12	0	38	0
I2	15/11/2007	0	3803	0	1817	0	0	0	0	0	1113	0	0	0	0	0	206	0
I2	13/12/2007	0	0	91278	1389102	0	0	0	0	0	0	0	0	0	0	0	5768	0
I2	17/01/2008	338	1981	634	8388	0	0	0	0	0	2782	0	549	1769	0	0	1053	0
I2	19/02/2008	368	0	22776	0	84	0	0	0	0	0	0	0	0	0	0	150	0
I2	18/03/2008	0	0	0	354322	35700	0	0	0	0	128195	0	0	0	24097	0	1007306	115700
I3	22/03/2005	0	0	0	17049	36892	0	0	0	0	0	0	1595	15408	0	736	0	0
I3	15/04/2005	0	0	0	763522	19833	0	0	246414	0	0	0	0	0	53550	43273	1179780	0
I3	11/05/2005	0	0	0	2510	87	0	0	0	0	0	0	0	1985	59	142	0	0
I3	09/06/2005	0	0	0	2227	264	0	0	0	0	0	0	18751	0	1904	0	0	0
I3	06/07/2005	0	0	0	15728	232	0	0	0	0	835	0	2472	0	471	380	2369	0
I3	01/08/2005	0	48899	0	3081	0	0	718	0	0	0	0	2623	1300	0	466	0	0
I3	31/08/2005	0	0	22919	5573	0	0	0	67	0	0	0	509	0	0	0	0	0
I3	04/10/2005	0	35576	0	25145	0	0	623	184	0	0	651	2803	0	0	0	0	0
I3	01/11/2005	0	0	0	6655	0	0	0	0	0	0	0	0	0	996	2414	0	0
I3	24/11/2005	0	1217	0	268	0	0	0	0	0	0	0	1582	0	402	0	0	0
I3	21/12/2005	0	45	0	88	1	0	0	0	0	0	0	0	124	0	0	0	0
I3	17/01/2006	0	0	0	25	7	0	0	0	0	0	91	98	314	0	0	0	0
I3	15/02/2006	0	0	0	5083	352	0	0	0	0	0	0	4993	32154	0	0	0	0
I3	15/03/2006	0	0	0	188761	0	0	0	0	0	0	0	0	7704	456	0	0	0
I3	20/04/2006	0	0	2645	1896	323	0	0	0	0	0	0	2292	0	0	0	0	0
I3	18/05/2006	0	0	79	192	15	0	0	0	0	0	64	0	442	0	0	0	0
I3	15/06/2006	0	0	0	47	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	13/07/2006	0	0	0	2045	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	10/08/2006	0	105540	0	66547	310	0	0	0	0	0	4085	0	0	418	15213	0	0
I3	07/09/2006	0	697264	0	441784	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	04/10/2006	0	0	384464	20703	0	0	0	0	0	0	0	0	5136	0	0	0	0
I3	02/11/2006	0	0	8052	1370	0	0	0	0	0	303	0	0	0	0	0	0	0
I3	30/11/2006	0	860	1229	515	0	0	0	0	0	54	50	107	0	81	0	0	0
I3	20/12/2006	0	396	440	297	0	44	0	0	0	35	0	0	332	0	32	0	0

Lake	Date	Asterionella sp.	Aulacoseira individuals	Small centric diatoms	Large centric diatoms	Synecra sp.	Cymbella sp.	Diatoma sp.	Fragilaria sp.	Gyrosigma sp.	Meridion sp.	Nitzschia sp.	Cocconeis	Navicula sp.	Small pennate diatoms	Tabellaria	Green Flagellate	Actinastrum hantzschii
I3	24/01/2007	0	0	4583	865	4	0	0	0	151	59	0	0	566	0	81	0	0
I3	15/02/2007	0	29209	178500	1381855	0	0	0	0	0	0	0	0	0	4819	5842	0	0
I3	15/03/2007	0	606681	0	2588	0	0	532	79	0	1212	0	0	7704	0	0	0	0
I3	19/04/2007	0	51405	0	13421	138	0	0	0	0	0	0	0	6289	0	0	3662	0
I3	17/05/2007	0	93926	1104	3167	0	0	0	0	0	0	0	957	0	0	0	120	0
I3	14/06/2007	0	93141	0	266234	2732	0	0	1697	0	0	0	0	207932	0	0	15068	0
I3	16/07/2007	0	2028	10142	5872	0	0	0	0	0	0	0	0	0	0	0	1694	0
I3	08/08/2007	0	0	80325	547588	0	0	0	0	0	0	0	0	0	0	20446	791	0
I3	11/09/2007	0	273835	774039	328553	12495	0	0	0	0	0	0	0	81509	0	110994	10020	0
I3	15/10/2007	67	2021	4043	4124	0	0	0	0	0	444	0	0	0	0	202	1314	0
I3	15/11/2007	0	3962	634	5452	0	0	0	0	0	0	0	0	0	0	0	206	0
I3	13/12/2007	0	769	0	373	0	0	0	16	0	185	0	0	1179	12	0	31	0
I3	17/01/2008	0	0	158	629	2	0	0	0	0	209	511	275	0	0	0	87	0
I3	19/02/2008	0	0	0	1458446	826	0	0	0	0	0	0	0	0	0	0	1465	0
I3	18/03/2008	0	0	0	278304	64260	0	0	0	0	0	0	0	65208	0	0	44722	0

Lake	Date	Ankistrodesmus falcatus	Ankyra ancora	Ankyra judyaii	Chlamydomonas spp.	Chlorella spp.	Chlorococcus spp.	Closterium sp.	Closterium acutum var. variable	Coelastrum microporum	Cosmarium sp.	Crucigeniella rectangularis	Dictyosphaerium pulchellum	Eudorina elegans	Lagerheimia sp.	Micratinium pusillum	Monoraphidium contortum	Pediastrum sp.	Pediastrum boryanum v. longicorne
C1	22/03/2005	0	0	0	0	9786	0	0	0	0	0	0	0	0	0	0	1067	0	0
C1	15/04/2005	0	0	0	0	25326	0	18546	0	205604	0	0	96431	0	0	0	6350	0	0
C1	11/05/2005	0	0	1275	0	12922	0	0	0	31471	0	0	0	0	0	0	211	0	7925
C1	09/06/2005	0	0	3408	0	4380	0	0	0	2052	0	0	0	0	0	0	0	173	0
C1	06/07/2005	0	0	213	10151	10337	0	0	0	62943	0	345	0	0	0	0	113	0	7925
C1	01/08/2005	63106	0	91819	381340	0	0	408791	0	10574346	0	49718	354250	0	0	0	0	0	0
C1	31/08/2005	9015	0	61213	0	0	0	408791	0	0	0	0	0	0	0	0	2028	0	0
C1	04/10/2005	0	0	0	46695	0	0	0	0	0	0	0	0	0	0	0	10432	0	0
C1	01/11/2005	0	0	2449	71183	0	0	0	0	0	0	0	0	0	0	0	1460	0	10144
C1	24/11/2005	0	0	0	2965	0	0	0	0	0	0	0	0	0	0	0	108	0	0
C1	21/12/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	129	0	0
C1	17/01/2006	0	0	0	772	0	0	0	0	0	0	0	0	0	0	0	176	0	0
C1	15/02/2006	0	0	0	32283	0	0	0	0	0	0	0	0	0	0	0	4894	0	0
C1	15/03/2006	0	0	0	5775	0	0	0	0	0	0	0	0	0	0	0	2843	39612	0
C1	20/04/2006	0	0	0	81941	0	0	0	0	299627	0	0	0	0	0	0	1810	0	0
C1	18/05/2006	0	0	877	1159	162	0	0	0	0	0	0	0	0	0	0	132	0	0
C1	15/06/2006	0	1230	260	541	429	0	0	0	803	0	0	0	0	0	0	0	0	0
C1	13/07/2006	0	0	9172	9524	74351	0	0	0	1584501	3262	29800	106164	0	0	0	608	0	76000
C1	10/08/2006	8013	0	81617	0	27566	0	0	0	8728031	0	22097	0	0	0	0	0	0	0
C1	07/09/2006	0	0	0	0	1211	0	1331	0	17703	0	97	0	0	0	0	4	0	0
C1	04/10/2006	0	0	0	0	0	0	17033	0	188828	0	3107	0	0	0	0	380	0	0
C1	02/11/2006	0	0	0	0	1184	0	0	0	2623	0	0	0	0	0	0	46	0	0
C1	30/11/2006	0	0	159	0	65	0	0	0	0	0	0	0	0	0	0	3	0	0
C1	20/12/2006	0	0	39	101	7	0	0	0	0	0	0	0	0	0	0	0	0	0
C1	24/01/2007	0	0	0	667	0	0	68	0	0	0	0	0	0	0	0	3	0	0
C1	15/02/2007	0	0	1853	0	4684	0	0	0	1967	0	0	0	0	0	0	0	0	83
C1	15/03/2007	0	0	0	222448	0	0	0	0	251770	0	0	0	0	0	0	4057	0	0
C1	19/04/2007	0	0	0	0	2016	0	0	0	2518	0	0	148	0	0	0	1	0	53
C1	17/05/2007	0	0	4251	0	323	0	237	0	0	0	0	0	0	0	0	0	0	0
C1	14/06/2007	0	0	0	205924	223287	0	0	0	543823	0	0	0	45485	0	0	730	0	0
C1	16/07/2007	0	0	0	23347	22784	0	50056	0	832383	0	0	0	0	0	0	4843	0	0
C1	08/08/2007	1082	0	0	0	22329	0	0	0	181274	0	0	0	0	0	30718	730	0	0
C1	11/09/2007	0	0	22486	46695	182275	0	100112	0	2219688	0	36528	0	0	0	0	5961	0	0
C1	15/10/2007	0	0	3826	0	0	0	34066	0	94414	0	0	0	0	0	0	127	0	0
C1	15/11/2007	0	679	0	1324	3877	0	8516	0	31471	0	0	0	0	0	0	1183	0	0
C1	13/12/2007																		
C1	17/01/2008	0	0	0	62	30	0	0	0	369	0	0	0	0	0	0	1	0	0
C1	19/02/2008	0	0	0	10593	0	0	0	0	0	0	0	0	0	0	0	0	0	0
C1	18/03/2008	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	12170	0	0
C2	22/03/2005	0	2394	0	11674	9114	0	5006	0	55492	0	0	0	0	0	0	4843	0	0
C2	15/04/2005	899	0	0	0	0	0	0	0	75322	0	0	17663	0	0	0	4855	0	0
C2	11/05/2005	0	0	0	0	387	0	0	0	0	0	0	0	0	0	0	51	0	263
C2	09/06/2005	0	0	24994	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
C2	06/07/2005	0	0	0	34321	5582	0	12264	0	271912	0	0	0	0	0	0	2008	0	45647
C2	01/08/2005	64373	0	0	0	0	0	0	0	19775399	0	0	1264759	0	0	0	28968	0	0
C2	31/08/2005	0	0	110183	0	0	0	0	0	3262941	0	0	0	0	0	0	0	45777	0
C2	04/10/2005	0	0	0	0	0	0	0	0	4028322	0	0	0	0	0	0	13523	0	0
C2	01/11/2005	0	0	0	0	0	0	3020	0	0	0	0	0	0	0	0	0	0	0
C2	24/11/2005	0	0	1302	0	0	0	3477	0	0	0	0	0	0	0	0	535	0	0
C2	21/12/2005	0	0	85	0	0	0	0	0	0	0	0	0	0	0	0	135	0	0
C2	17/01/2006	0	0	0	953	0	0	0	0	2831	0	0	0	0	0	0	243	0	0
C2	15/02/2006	0	0	0	0	0	0	0	0	52950	0	0	0	0	0	0	510	0	0
C2	15/03/2006	0	0	956	0	0	0	0	0	47207	0	0	5535	0	0	0	1616	0	0
C2	20/04/2006	894	0	0	315157	0	0	0	0	0	0	0	105397	0	0	0	1609	0	0
C2	18/05/2006	34	0	2488	841	234	0	0	0	0	0	0	0	0	0	0	27	0	240
C2	15/06/2006	0	0	1963	7115	270	0	0	0	0	0	0	0	0	0	0	134	0	276
C2	13/07/2006	0	0	26079	379084	0	0	116106	0	5148625	0	0	0	0	0	0	17283	108348	0
C2	10/08/2006	0	0	176293	549130	0	0	0	0	4350588	0	0	0	0	0	0	11684	0	0
C2	07/09/2006	0	0	0	68641	44657	0	0	0	1087647	0	0	127530	0	0	0	1460	0	45647
C2	04/10/2006	0	0	9794	0	39695	0	0	0	0	0	0	0	0	0	0	325	142417	0
C2	02/11/2006	0	0	285	5318	0	0	422	0	0	0	0	0	0	0	0	94	0	0
C2	30/11/2006	0	0	0	0	215	0	0	0	0	0	0	0	0	0	0	2	147	0
C2	20/12/2006	0	0	50	1626	0	0	0	0	0	0	27	0	0	0	0	6	0	0
C2	24/01/2007	0	0	0	0	0	0	38	0	211	0	0	0	106	0	0	0	0	0
C2	15/02/2007	0	0	0	160	0	0	0	0	1904	0	0	0	0	0	0	0	0	0
C2	15/03/2007	0	0	0	17430	0	0	3397	0	0	0	0	0	0	0	0	607	0	0
C2	19/04/2007	0	0	0	0	10660	0	0	0	61369	0	0	0	395	0	267	38	0	1189
C2	17/05/2007	0	0	4493	8483	4967	0	0	0	0	0	0	0	0	0	0	36	0	564
C2	14/06/2007	0	0	4497	438930	9114	0	20022	0	0	0	0	0	18565	0	12538	1192	0	0
C2	16/07/2007	0	0	0	10169	6616	0	7267	0	241699	0	2652	0	0	0	0	1190	0	0
C2	08/08/2007	0	0	689	0	9769	0	0	0	33989	0	0	0	0	0	0	639	0	0
C2	11/09/2007	0	0	0	0	0	0	0	0	849724	0	0	0	0	0	0	10269	0	0
C2	15/10/2007	0	0	0	0	31012	0	0	0	283241	0	0	22141	0	0	0	887	0	0

Lake	Date	Ankistrodesmus falcatus	Ankyra ancora	Ankyra judyaii	Chlamydomonas spp.	Chlorella spp.	Chlorococcus spp.	Closterium sp.	Closterium acutum var. variable	Coelastrum microporum	Cosmarium sp.	Crucigeniella rectangularis	Dictyosphaerium pulchellum	Eudorina elegans	Lagerheimia sp.	Micratinium pusillum	Monoraphidium contortum	Pediastrum sp.	Pediastrum boryanum v. longicorne
C2	15/11/2007	0	0	0	993	969	0	0	0	23603	0	0	0	0	0	0	222	0	0
C2	13/12/2007	0	0	15	0	0	0	0	0	738	0	0	0	0	0	0	7	0	0
C2	17/01/2008	0	0	6	0	0	0	0	0	291	0	0	0	0	0	0	1	0	0
C2	19/02/2008	0	0	0	13241	15506	0	0	0	0	0	0	0	0	0	0	254	0	0
C2	18/03/2008	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	649	0	0
C3	22/03/2005	530	0	0	0	0	0	16018	0	88788	0	0	0	0	0	0	3338	0	0
C3	15/04/2005	0	0	0	0	381051	0	81673	0	452715	0	0	106164	0	0	0	32826	0	0
C3	11/05/2005	0	0	582	0	1180	0	648	0	93403	0	0	0	0	0	0	0	0	0
C3	09/06/2005	0	0	211551	0	0	0	0	0	870118	0	0	0	0	0	0	0	0	0
C3	06/07/2005	0	0	0	0	0	497	0	1970	11098	0	0	0	0	0	0	0	1868	0
C3	01/08/2005	0	0	0	460786	0	0	34066	0	1510621	0	0	708499	0	0	0	13692	0	0
C3	31/08/2005	0	0	179891	0	182275	0	0	787915	48833129	0	0	4164241	0	0	0	0	0	0
C3	04/10/2005	0	0	79951	0	20253	0	0	612823	493264	0	0	231347	0	0	0	0	0	0
C3	01/11/2005	481	0	1088	0	1103	0	0	26215	53711	0	0	62978	0	0	0	0	4521	0
C3	24/11/2005	0	0	0	0	0	0	0	53587	0	0	0	59624	0	0	0	0	0	0
C3	21/12/2005	0	0	379	394	0	0	0	0	3121	0	0	0	0	0	0	180	0	0
C3	17/01/2006	0	0	0	0	0	0	0	0	11802	0	0	0	0	0	0	951	0	0
C3	15/02/2006	0	0	3212	0	0	0	0	0	0	0	0	0	0	0	0	31295	0	0
C3	15/03/2006	0	0	0	9269	0	0	0	0	0	0	0	0	0	0	0	2282	0	0
C3	20/04/2006	0	0	0	11674	0	0	0	0	277461	0	0	0	0	0	0	10804	0	0
C3	18/05/2006	0	0	1205	36	0	0	0	0	862	0	0	0	0	0	0	0	0	0
C3	15/06/2006	0	0	1842	147	287	0	0	0	0	0	0	0	0	0	0	0	0	0
C3	13/07/2006	0	47887	0	186779	45569	0	0	0	2219688	0	0	0	0	0	0	7451	0	0
C3	10/08/2006	0	113669	0	63337	30905	0	0	0	5268878	0	0	0	0	0	0	5053	0	0
C3	07/09/2006	0	0	0	0	0	0	0	0	2219688	0	0	0	0	0	0	5961	0	0
C3	04/10/2006	0	0	7625	0	23179	0	16974	0	376348	0	0	0	0	0	0	505	0	0
C3	02/11/2006	0	0	0	0	1108	0	0	0	0	0	0	0	1128	0	0	290	0	0
C3	30/11/2006	0	0	120	0	0	0	0	0	0	0	0	0	0	0	0	32	0	0
C3	20/12/2006	0	0	60	0	0	0	0	0	1475	0	0	0	0	0	0	5	0	0
C3	24/01/2007	0	0	19	0	434	0	87	0	2880	0	0	0	0	0	0	8	0	0
C3	15/02/2007	0	0	80	0	0	0	0	0	0	0	0	0	0	0	0	5	0	0
C3	15/03/2007	0	0	0	11864	3308	0	3634	0	0	0	0	0	0	0	0	865	0	0
C3	19/04/2007	0	509	0	0	19383	0	2129	0	106216	0	0	0	0	0	0	32	993	0
C3	17/05/2007	0	0	717	11420	7753	0	0	0	11802	0	0	0	0	0	0	349	993	0
C3	14/06/2007	0	0	0	38134	49619	0	0	0	0	0	0	0	0	0	0	3245	76295	0
C3	16/07/2007	0	0	0	18643	0	0	3634	0	281983	0	0	18893	0	0	0	703	0	0
C3	08/08/2007	0	0	0	7503	0	0	0	0	0	0	0	0	877	0	0	155	0	0
C3	11/09/2007	1123	0	0	23751	30905	0	16974	0	188174	0	0	44128	0	0	0	1011	0	0
C3	15/10/2007	0	0	1913	3972	1938	0	42582	0	613690	0	0	0	0	0	0	317	0	0
C3	15/11/2007	0	0	1874	0	11392	0	8343	0	0	0	0	0	0	0	0	3105	0	0
C3	13/12/2007	0	16	0	191	0	0	136	0	0	0	0	0	0	0	0	12	0	0
C3	17/01/2008	0	0	26	0	1034	0	114	0	3777	0	0	295	0	0	0	3	0	0
C3	19/02/2008	0	0	0	10593	0	0	0	0	0	0	0	0	0	0	0	1352	0	0
C3	18/03/2008	0	0	34432	0	0	0	0	0	0	0	0	0	0	0	0	3423	0	0
I1	22/03/2005	0	3721	0	0	0	0	0	0	0	0	0	0	0	0	0	21	0	0
I1	15/04/2005	0	0	19	0	38	0	0	0	0	0	0	215	0	0	0	0	0	0
I1	11/05/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	28	193	0
I1	09/06/2005	0	0	187	0	1134	0	0	0	0	0	0	393	0	0	0	39	70	0
I1	06/07/2005	0	0	0	2317	323	0	0	0	11802	0	0	0	0	0	0	0	0	0
I1	01/08/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5	0	0
I1	31/08/2005	0	36	0	0	208	0	0	0	843	36	0	0	141	0	0	18	213	0
I1	04/10/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I1	01/11/2005	0	0	125	130	95	0	0	0	0	0	0	90	0	0	0	1	32	0
I1	24/11/2005	0	0	0	95	0	0	0	0	0	0	0	0	0	0	0	0	64	0
I1	21/12/2005	0	0	0	0	35	0	0	0	0	0	0	0	71	0	0	0	0	0
I1	17/01/2006	0	0	27	0	0	4	0	0	0	0	0	246	44	0	0	0	0	0
I1	15/02/2006	0	0	0	865	0	0	0	0	0	0	0	2411	2579	0	0	587	433	0
I1	15/03/2006	0	0	0	0	0	0	0	0	0	0	0	0	4299	0	0	0	0	0
I1	20/04/2006	0	0	0	17279	0	0	0	0	0	0	1709	0	33956	0	0	0	0	0
I1	18/05/2006	0	0	8799	43695	775	0	0	0	0	0	0	0	0	0	0	0	0	0
I1	15/06/2006	0	0	0	9122	574	0	0	0	0	0	0	0	0	0	0	188	0	0
I1	13/07/2006	0	0	0	33847	0	0	0	0	0	0	0	0	26914	0	0	2809	0	0
I1	10/08/2006	0	0	0	9931	1938	0	0	0	47207	0	0	0	59225	0	13333	254	0	0
I1	07/09/2006	0	0	0	115196	3877	0	0	0	94414	0	0	0	75019	0	10666	127	0	0
I1	04/10/2006	0	0	0	17875	4361	0	0	0	106216	0	0	0	0	0	0	111	993	0
I1	02/11/2006	0	0	0	31668	1932	0	0	0	70565	0	0	0	0	0	0	0	0	1974
I1	30/11/2006	0	0	0	1036	126	0	0	0	0	0	0	0	0	0	0	8	0	0
I1	20/12/2006	0	0	8	50	0	0	0	0	799	0	0	0	0	0	0	0	0	0
I1	24/01/2007	0	0	68	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I1	15/02/2007																		
I1	15/03/2007	0	0	4997	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2070
I1	19/04/2007	0	0	2451	186	61	0	0	0	0	0	0	0	0	0	0	0	0	0
I1	17/05/2007	0	0	2366	1228	52	0	0	0	2538	0	0	0	0	0	0	0	0	0

Lake	Date	Ankistrodesmus falcatus	Ankyra ancora	Ankyra judyaii	Chlamydomonas spp.	Chlorella spp.	Chlorococcus spp.	Closterium sp.	Closterium acutum var. variable	Coelastrum microporum	Cosmarium sp.	Crucigeniella rectangularis	Dictyosphaerium pulchellum	Eudorina elegans	Lagerheimia sp.	Micratinium pusillum	Monoraphidium contortum	Pediastrum sp.	Pediastrum boryanum v. longicorne
I1	14/06/2007	0	0	598	7820	3271	0	266	0	0	0	0	0	0	0	0	103	0	0
I1	16/07/2007	0	0	438	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
I1	08/08/2007	0	0	1033	715	0	0	0	0	0	0	0	0	0	0	0	46	0	0
I1	11/09/2007	0	0	0	497	0	0	0	0	0	0	0	0	0	0	0	16	993	0
I1	15/10/2007	0	0	123	0	83	0	0	0	0	0	0	0	0	0	0	0	0	0
I1	15/11/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I1	13/12/2007	0	0	0	497	0	0	0	0	11802	0	0	0	0	0	0	254	0	0
I1	17/01/2008	0	0	120	0	363	0	0	0	0	0	0	0	0	0	0	4	0	0
I1	19/02/2008	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3138	0	0
I1	18/03/2008	0	0	3188	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	22/03/2005	0	0	0	0	0	0	0	0	0	0	0	2285	0	0	0	39	0	0
I2	15/04/2005	0	0	413	0	0	0	0	0	3395	0	0	0	0	0	0	91	0	0
I2	11/05/2005	0	365	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	09/06/2005	0	0	3415	67365	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	06/07/2005	43	0	0	0	30403	0	648	0	0	0	0	0	0	0	0	0	0	0
I2	01/08/2005	0	0	60	1738	0	0	0	0	5901	0	0	45665	0	0	0	0	0	0
I2	31/08/2005	0	0	0	0	969	0	4258	0	23603	0	0	5535	0	0	0	0	0	0
I2	04/10/2005	0	0	461	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	01/11/2005	0	0	115	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	24/11/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0
I2	21/12/2005	0	0	120	0	0	0	0	0	0	0	0	0	0	0	0	20	0	0
I2	17/01/2006	0	0	179	0	0	0	0	0	0	0	0	0	0	0	0	190	0	0
I2	15/02/2006	0	0	0	0	0	0	1065	0	11802	0	0	0	0	0	0	317	993	0
I2	15/03/2006	0	0	0	0	1938	0	0	0	0	0	0	0	0	0	0	63	0	0
I2	20/04/2006	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	127	0	0
I2	18/05/2006	0	0	56840	9080	0	0	0	0	0	0	0	0	0	0	0	290	0	0
I2	15/06/2006	0	0	0	65	0	0	0	0	0	0	0	361	0	11	0	0	0	0
I2	13/07/2006	0	0	521	0	0	0	0	0	25701	0	0	0	0	0	0	0	0	0
I2	10/08/2006	0	0	0	7150	0	0	1703	0	113297	0	0	26569	0	0	3200	76	0	0
I2	07/09/2006	0	0	48	0	169	0	424	0	2937	0	0	0	0	0	0	0	0	0
I2	04/10/2006	0	0	393	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	02/11/2006	0	0	166	0	898	0	0	0	2735	0	0	12829	0	0	0	0	0	0
I2	30/11/2006	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	9	0	0
I2	20/12/2006	0	0	0	47	0	0	0	0	0	0	0	0	0	0	0	9	0	0
I2	24/01/2007	0	0	0	353	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	15/02/2007																		
I2	15/03/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	19/04/2007	0	6393	0	0	0	0	237	0	0	0	0	1845	0	0	0	0	0	0
I2	17/05/2007	0	0	7014	0	0	0	0	0	20981	0	0	0	0	0	0	56	0	0
I2	14/06/2007	7	0	520	2207	191	0	0	0	1166	0	0	0	0	0	0	0	0	0
I2	16/07/2007	0	0	850	9269	1723	0	0	0	0	0	0	0	877	0	0	408	0	0
I2	08/08/2007	0	0	1063	2648	8184	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	11/09/2007	0	0	691	0	108	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	15/10/2007	0	0	877	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0
I2	15/11/2007	0	0	1674	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	13/12/2007	0	0	8608	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	17/01/2008	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	983	0	0
I2	19/02/2008	0	0	3124	0	0	0	0	0	0	0	0	0	0	0	0	52	0	0
I2	18/03/2008	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	434484	0	0
I3	22/03/2005	0	0	694	10092	0	0	0	0	0	0	0	0	0	0	0	506	0	0
I3	15/04/2005	0	0	40809	169485	0	0	0	0	0	0	0	0	0	0	0	148750	0	0
I3	11/05/2005	0	0	134	0	1087	0	0	0	0	0	0	0	0	0	0	0	0	1111
I3	09/06/2005	0	0	0	0	1103	0	0	0	53711	0	0	0	0	0	0	0	0	4508
I3	06/07/2005	0	0	6815	1490	9449	0	1597	0	17703	0	0	4151	0	0	0	214	0	0
I3	01/08/2005	0	0	0	0	0	0	0	0	8671	0	0	0	0	0	0	29	365	0
I3	31/08/2005	0	0	0	0	0	0	0	0	98474	0	360	0	0	0	618	132	0	3674
I3	04/10/2005	0	0	0	633	0	0	0	0	0	0	0	14121	0	0	0	546	0	0
I3	01/11/2005	0	0	1138	2364	0	0	0	0	56180	0	0	0	0	0	0	0	4729	0
I3	24/11/2005	0	0	689	5720	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	21/12/2005	0	0	8	105	17	0	75	0	0	0	0	0	0	0	0	7	0	0
I3	17/01/2006	0	0	128	265	258	0	0	0	0	0	0	0	0	0	0	175	0	0
I3	15/02/2006	0	0	0	0	0	0	0	0	214526	0	0	0	0	0	0	0	0	0
I3	15/03/2006	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2347	0	0
I3	20/04/2006	0	0	499	0	0	0	0	0	0	0	0	0	0	0	0	1240	0	0
I3	18/05/2006	0	0	7114	621	121	0	0	0	0	0	0	692	0	0	0	0	0	0
I3	15/06/2006	0	0	3108	8358	4280	0	0	0	0	0	0	461	0	0	0	0	0	0
I3	13/07/2006	0	0	48742	3631	9302	0	973	0	0	0	0	0	0	0	0	29	0	4528
I3	10/08/2006	0	0	478	993	0	0	4258	0	94414	0	0	0	0	0	5333	222	0	5944
I3	07/09/2006	0	0	0	0	11630	0	0	0	0	0	0	0	0	0	0	380	0	0
I3	04/10/2006	0	0	0	2883	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	02/11/2006	0	0	260	0	791	0	0	0	0	0	0	0	0	0	0	26	0	0
I3	30/11/2006	0	0	139	0	0	0	0	0	1144	0	0	0	0	0	0	0	96	0
I3	20/12/2006	0	0	942	31	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Lake	Date	Ankistrodesmus falcatus	Ankyra ancora	Ankyra judyaii	Chlamydomonas spp.	Chlorella spp.	Chlorococcus spp.	Closterium sp.	Closterium acutum var. variable	Coelastrum microporum	Cosmarium sp.	Crucigeniella rectangularis	Dictyosphaerium pulchellum	Eudorina elegans	Lagerheimia sp.	Micratinium pusilum	Monoraphidium contortum	Pediastrum sp.	Pediastrum boryanum v. longicorne
I3	24/01/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	10	212	0
I3	15/02/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	365	0	0
I3	15/03/2007	0	0	0	541	0	0	0	0	0	0	0	0	0	0	0	190	0	0
I3	19/04/2007	0	0	0	1765	10337	0	0	0	41962	0	0	0	0	0	0	2874	0	0
I3	17/05/2007	0	0	0	0	13509	0	0	0	20563	0	0	4822	0	0	0	55	0	0
I3	14/06/2007	0	0	44973	0	11392	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	16/07/2007	0	0	1435	0	3877	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	08/08/2007	0	0	2755	125842	0	0	0	0	0	0	0	0	0	0	0	365	11444	0
I3	11/09/2007	0	0	0	0	44657	0	0	0	0	0	0	0	0	0	0	1826	0	22824
I3	15/10/2007	0	0	381	1188	6181	0	1697	0	84678	0	0	0	0	0	0	51	0	0
I3	15/11/2007	0	0	0	0	485	0	0	0	0	0	0	0	0	0	0	16	0	0
I3	13/12/2007	0	0	0	55	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	17/01/2008	0	0	179	0	0	0	0	0	0	0	0	0	0	0	0	5	0	0
I3	19/02/2008	0	0	10202	0	0	0	22711	0	0	0	0	0	0	0	0	338	0	0
I3	18/03/2008	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	19862	0	0

Lake	Date	Pediastrum duplex	Pediastrum simplex	Pteromonas sp.	Quadrigula sp.	Scenedesmus abundans	Scenedesmus arcuatus	Scenedesmus bicaudatus	Scenedesmus communis (2 cell)	Scenedesmus communis (4 cell)	Scenedesmus communis (8 cell)	Scenedesmus falcatus (2 cell)	Scenedesmus falcatus (4 cell)	Scenedesmus falcatus (8 cell)	Scenedesmus opoliensis (4 cell)	Scenedesmus opoliensis (2 cell)	Selenastrum sp.	Schroederia sp.	Staurastrum punctulatum
C1	22/03/2005	0	0	0	0	0	0	0	740	1482	0	0	2880	0	0	0	0	0	0
C1	15/04/2005	0	0	0	0	0	0	0	7657	38344	0	0	7454	0	0	0	0	0	0
C1	11/05/2005	0	0	0	0	0	0	0	879	3522	0	0	0	0	5788	0	0	0	0
C1	09/06/2005	0	0	0	0	0	0	0	0	38	0	0	0	0	0	0	0	0	0
C1	06/07/2005	0	0	0	0	0	0	0	293	391	0	0	0	0	0	0	0	0	0
C1	01/08/2005	0	0	0	0	0	0	0	28127	140861	0	0	1697775	0	764078	0	0	0	0
C1	31/08/2005	0	0	0	0	0	0	0	253145	929686	0	54722	2081143	0	0	0	0	0	0
C1	04/10/2005	0	0	0	0	0	0	0	268644	434658	0	120611	885212	0	0	0	0	0	0
C1	01/11/2005	0	0	0	0	0	0	0	23627	72121	0	0	35051	0	0	0	0	0	16454
C1	24/11/2005	0	0	0	0	0	0	0	328	282	0	0	1095	0	0	0	57	0	0
C1	21/12/2005	0	0	0	0	0	0	0	424	749	0	0	680	0	493	0	0	0	0
C1	17/01/2006	0	0	0	0	0	0	0	293	587	0	48	380	0	0	0	0	59	0
C1	15/02/2006	0	0	0	0	0	0	0	893	7155	0	1737	0	0	0	0	0	0	0
C1	15/03/2006	0	0	0	0	0	0	0	365	366	0	0	0	0	0	901	0	0	0
C1	20/04/2006	0	0	0	0	0	0	0	0	0	0	0	5431	0	13778	0	0	0	0
C1	18/05/2006	0	0	0	0	0	0	0	73	147	0	71	285	0	724	0	0	0	0
C1	15/06/2006	415	0	0	0	0	0	0	15	45	0	0	0	0	74	18	0	0	0
C1	13/07/2006	116958	0	0	0	0	0	0	12644	67543	0	0	41032	16413	10408	0	5133	0	0
C1	10/08/2006	231268	0	0	0	0	0	0	143761	575967	25042	12160	584181	389454	710052	46278	15226	0	30471
C1	07/09/2006	0	0	0	0	0	0	0	632	1541	0	0	535	0	950	0	201	0	0
C1	04/10/2006	0	0	0	0	0	0	0	7032	12325	0	0	6846	0	13024	0	0	0	0
C1	02/11/2006	0	0	0	0	0	0	0	317	196	0	0	95	0	1085	482	30	0	119
C1	30/11/2006	90	0	0	0	0	0	0	10	59	0	0	19	0	48	0	0	0	0
C1	20/12/2006	0	0	0	0	0	0	0	9	9	0	0	0	0	30	11	0	0	0
C1	24/01/2007	0	0	0	0	0	0	0	127	85	0	14	27	0	278	87	0	0	0
C1	15/02/2007	0	0	0	0	0	0	0	64	220	37	0	0	0	0	0	0	0	0
C1	15/03/2007	0	0	0	0	0	0	0	0	0	0	4560	0	0	0	0	0	0	0
C1	19/04/2007	0	0	0	0	0	0	0	29	82	23	0	0	0	87	0	0	0	0
C1	17/05/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	118	0
C1	14/06/2007	0	0	0	0	0	0	0	0	50710	0	0	0	0	0	0	0	0	0
C1	16/07/2007	0	0	0	0	0	0	0	2583	36222	0	5025	20118	20118	0	0	0	0	0
C1	08/08/2007	0	0	0	0	0	0	0	18564	33807	0	0	46004	39432	50012	0	0	0	0
C1	11/09/2007	0	0	0	0	0	0	0	51662	165584	0	20102	160948	80474	51033	0	0	0	0
C1	15/10/2007	0	7787	462	0	0	0	0	1758	1761	0	0	10269	6846	0	0	0	0	0
C1	15/11/2007	0	0	462	0	0	0	0	293	2348	0	0	1141	0	5788	0	0	0	0
C1	13/12/2007																		
C1	17/01/2008	0	0	0	0	0	0	0	10	28	0	13	13	0	0	0	0	0	0
C1	19/02/2008	0	0	0	0	0	0	0	0	0	0	0	9128	0	0	0	0	0	0
C1	18/03/2008	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
C2	22/03/2005	0	0	0	0	0	0	0	0	3105	0	0	0	0	0	0	629	0	0
C2	15/04/2005	0	0	0	0	0	0	0	0	8428	0	0	13654	0	0	0	0	0	0
C2	11/05/2005	0	0	0	0	0	0	0	0	468	0	0	0	0	0	0	0	0	0
C2	09/06/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
C2	06/07/2005	0	0	0	0	0	0	0	1266	7607	0	0	4929	0	0	0	0	0	0
C2	01/08/2005	0	0	0	0	0	24730	0	100421	771129	0	65124	1499075	651772	0	0	0	0	163184
C2	31/08/2005	0	0	0	0	0	0	0	111384	477892	0	0	571767	0	0	0	12333	0	0
C2	04/10/2005	0	0	0	0	0	0	0	300024	788824	0	255369	1314406	0	0	0	0	0	0
C2	01/11/2005	0	0	0	0	0	460	0	15583	33089	0	0	15778	0	0	0	0	0	0
C2	24/11/2005	0	0	0	0	0	0	0	2034	4553	0	0	3261	932	0	0	0	0	0
C2	21/12/2005	0	0	0	0	0	0	0	1246	2184	0	151	910	0	0	0	0	0	0
C2	17/01/2006	0	0	0	0	0	0	0	764	1320	0	103	411	0	781	0	0	0	0
C2	15/02/2006	0	0	0	0	0	0	0	329	658	0	0	0	0	0	0	0	0	0
C2	15/03/2006	0	0	0	0	0	0	0	0	440	0	428	856	0	0	0	0	0	0
C2	20/04/2006	0	0	0	0	0	0	0	0	2794	0	0	0	0	0	0	10192	0	0
C2	18/05/2006	0	0	0	0	0	0	0	80	107	0	0	0	0	0	0	0	0	0
C2	15/06/2006	0	0	0	0	0	0	0	61	153	0	0	0	0	0	0	0	0	0
C2	13/07/2006	0	0	0	0	0	0	0	71899	624125	0	69940	933306	0	0	0	14595	0	0
C2	10/08/2006	0	0	0	0	0	0	0	465787	2271814	0	433397	1892745	0	0	0	0	0	0
C2	07/09/2006	0	0	0	0	0	0	0	65818	162272	0	19700	59148	0	0	0	0	0	0
C2	04/10/2006	0	0	0	0	0	0	0	13501	58598	0	0	8763	0	0	0	0	0	0
C2	02/11/2006	0	0	0	0	0	0	0	1700	4191	0	85	339	0	0	0	0	0	425
C2	30/11/2006	0	0	0	0	0	0	0	98	359	0	0	0	0	0	0	0	0	0
C2	20/12/2006	0	0	0	0	0	0	0	23	153	0	0	0	0	0	0	0	0	0
C2	24/01/2007	0	0	0	0	0	0	0	16	51	0	0	15	0	0	0	0	0	0
C2	15/02/2007	0	0	0	0	0	0	0	18	36	0	34	138	0	175	0	0	0	0
C2	15/03/2007	0	0	0	0	0	0	0	351	702	0	0	0	0	0	0	0	0	0
C2	19/04/2007	0	0	0	0	0	0	0	88	792	0	0	171	342	1519	0	0	0	0
C2	17/05/2007	0	0	0	0	0	0	0	0	251	501	0	244	0	309	0	0	0	0
C2	14/06/2007	0	0	0	0	0	0	0	0	4140	0	0	0	0	10207	0	0	0	0
C2	16/07/2007	0	0	0	0	0	0	0	750	21035	0	0	2921	0	14818	0	0	0	0
C2	08/08/2007	0	0	0	0	0	0	0	5379	6973	0	1231	3697	0	0	2343	0	0	0
C2	11/09/2007	0	0	0	0	0	0	0	47465	158469	0	0	123226	123226	39072	0	0	0	0
C2	15/10/2007	0	0	0	0	0	0	0	0	15847	0	0	6846	0	8683	0	0	0	0

Lake	Date	Pediastrum duplex	Pediastrum simplex	Pteromonas sp.	Quadrigula sp.	Scenedesmus abundans	Scenedesmus arcuatus	Scenedesmus bicaudatus	Scenedesmus communis (2 cell)	Scenedesmus communis (4 cell)	Scenedesmus communis (8 cell)	Scenedesmus falcatus (2 cell)	Scenedesmus falcatus (4 cell)	Scenedesmus falcatus (8 cell)	Scenedesmus opoliensis (4 cell)	Scenedesmus opoliensis (2 cell)	Selenastrum sp.	Schroederia sp.	Staurastrum punctulatum
C2	15/11/2007	0	0	58	0	0	0	0	549	660	0	0	1711	856	0	0	0	0	0
C2	13/12/2007	0	0	0	0	0	0	0	27	0	0	0	53	0	68	17	0	0	0
C2	17/01/2008	0	0	0	0	0	0	0	0	0	0	0	11	0	0	0	0	0	0
C2	19/02/2008	0	0	0	0	0	0	0	0	2348	0	0	0	0	0	0	0	0	0
C2	18/03/2008	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
C3	22/03/2005	0	0	0	0	0	0	0	827	828	0	0	0	0	0	0	9061	0	0
C3	15/04/2005	0	0	0	0	0	0	0	4215	18237	0	0	24619	0	0	0	12833	0	0
C3	11/05/2005	0	0	0	0	0	0	0	201	268	0	0	521	0	0	0	0	0	0
C3	09/06/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
C3	06/07/2005	0	0	0	0	0	0	0	103	1449	0	0	0	0	0	0	0	0	0
C3	01/08/2005	0	0	0	0	0	0	0	101961	232421	0	109444	492902	27383	0	0	29975	0	0
C3	31/08/2005	0	0	0	0	0	0	0	1363883	2579800	0	1125707	5472223	0	0	0	352363	0	0
C3	04/10/2005	0	0	0	0	0	0	0	307677	357294	0	89342	304012	0	0	0	111861	0	22387
C3	01/11/2005	0	0	0	0	392	0	0	19001	31052	0	2432	20446	0	0	0	2436	0	1219
C3	24/11/2005	0	0	0	0	0	0	0	1627	5228	0	0	1152	0	0	0	1441	0	0
C3	21/12/2005	0	0	0	0	0	0	0	1453	3609	0	0	1132	0	0	0	71	0	0
C3	17/01/2006	0	0	0	0	0	0	0	2088	7263	0	0	856	0	1085	0	134	0	0
C3	15/02/2006	0	0	0	0	0	0	0	10332	0	0	0	11496	11496	7290	0	0	0	0
C3	15/03/2006	0	0	0	0	0	0	0	1172	2348	0	570	1141	0	1447	723	0	0	0
C3	20/04/2006	0	0	0	0	0	0	0	7749	20698	10349	0	0	0	12758	0	0	0	12593
C3	18/05/2006	0	0	0	0	0	0	0	80	16	0	0	0	0	40	0	0	0	0
C3	15/06/2006	0	0	0	0	0	0	0	33	130	0	0	0	0	0	0	0	0	0
C3	13/07/2006	0	0	0	0	0	0	0	185984	765826	41396	40204	281659	0	204132	0	25169	0	50371
C3	10/08/2006	0	0	0	0	0	0	0	231247	926469	0	109065	1419013	0	34611	0	8535	0	0
C3	07/09/2006	0	0	0	0	0	0	0	557952	848618	0	0	321895	0	0	0	0	0	0
C3	04/10/2006	64819	0	0	0	0	0	0	87593	164040	0	0	88688	0	8653	0	0	0	0
C3	02/11/2006	1162	0	0	0	0	0	0	5651	9558	0	244	4890	7824	3721	0	0	0	1224
C3	30/11/2006	762	0	0	0	0	0	0	1318	2311	0	160	214	0	136	0	0	0	0
C3	20/12/2006	0	0	0	0	0	0	0	179	193	28	13	53	0	271	0	0	0	0
C3	24/01/2007	0	0	0	0	0	0	0	27	286	36	0	17	0	0	0	0	0	0
C3	15/02/2007	0	0	0	0	0	0	0	73	220	0	0	0	0	0	0	0	0	0
C3	15/03/2007	0	0	0	0	0	0	0	1125	3005	1503	0	1460	0	0	0	457	0	0
C3	19/04/2007	0	0	0	0	0	0	0	330	2861	880	0	0	856	7055	0	0	0	0
C3	17/05/2007	0	0	0	0	0	0	0	110	1100	0	0	0	0	0	0	0	0	0
C3	14/06/2007	0	0	0	0	0	0	0	2813	39441	0	0	0	21907	0	0	0	0	0
C3	16/07/2007	0	0	0	0	0	0	0	1500	6761	3005	0	0	2921	0	0	914	0	0
C3	08/08/2007	0	0	0	0	0	0	0	781	783	0	0	1521	761	482	0	0	0	0
C3	11/09/2007	0	46559	0	0	0	0	0	3504	17547	0	0	34111	0	34611	0	0	0	0
C3	15/10/2007	0	7787	0	0	0	0	0	1318	14086	1761	0	25672	0	0	0	0	0	2143
C3	15/11/2007	0	0	453	0	0	0	0	1722	32772	6899	0	30178	6706	0	0	0	0	0
C3	13/12/2007	0	0	0	0	0	0	0	28	14	0	41	82	110	0	0	0	0	0
C3	17/01/2008	0	104	0	0	0	0	0	105	493	0	0	0	91	116	0	0	0	0
C3	19/02/2008	0	0	0	0	0	0	0	0	4695	0	0	0	0	0	0	0	0	0
C3	18/03/2008	0	0	0	0	0	0	0	15822	31694	0	0	0	0	0	0	0	0	0
I1	22/03/2005	0	0	0	0	0	0	0	73	0	0	0	0	0	0	0	0	0	0
I1	15/04/2005	0	0	0	14	0	0	0	26	17	0	0	33	0	42	0	0	0	42
I1	11/05/2005	0	0	0	0	0	0	0	64	0	0	0	0	0	0	0	0	0	0
I1	09/06/2005	0	0	0	0	0	0	0	0	31	0	0	0	0	0	0	0	0	0
I1	06/07/2005	339	0	0	0	0	0	0	0	220	0	0	0	0	1447	181	0	0	179
I1	01/08/2005	0	0	19	0	0	0	0	37	0	0	0	0	0	362	0	0	0	0
I1	31/08/2005	0	0	4	0	0	0	0	0	31	63	0	0	0	969	0	0	0	0
I1	04/10/2005	0	0	0	0	0	0	0	0	0	0	0	122	0	0	0	0	0	0
I1	01/11/2005	0	0	0	0	0	0	0	11	22	0	7	28	0	0	0	0	0	0
I1	24/11/2005	0	0	0	0	0	0	0	0	14	0	0	0	0	0	0	0	0	0
I1	21/12/2005	0	0	0	13	0	0	0	0	16	0	0	0	0	0	0	0	0	0
I1	17/01/2006	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	260	0
I1	15/02/2006	0	0	0	0	0	0	0	0	96	0	0	0	0	0	0	0	12374	0
I1	15/03/2006	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2313	0
I1	20/04/2006	0	0	0	0	0	0	0	0	88	0	0	0	0	0	0	0	1062	0
I1	18/05/2006	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	850	0
I1	15/06/2006	0	0	0	0	0	0	0	33	261	0	0	0	0	0	0	0	79	476
I1	13/07/2006	0	0	0	0	0	0	0	0	27005	0	0	2917	11666	0	0	3649	0	0
I1	10/08/2006	0	0	0	0	0	0	0	0	18488	0	0	0	0	75974	0	0	0	8570
I1	07/09/2006	0	0	0	0	0	0	0	0	2641	0	0	0	0	6512	0	0	0	12855
I1	04/10/2006	0	0	0	0	0	0	0	330	5282	0	0	856	0	0	0	0	0	3749
I1	02/11/2006	2026	0	0	0	0	0	0	876	5264	0	0	0	0	0	0	0	0	4270
I1	30/11/2006	0	0	0	0	0	0	0	29	115	0	0	0	0	0	0	0	0	0
I1	20/12/2006	0	0	0	0	0	0	43	4	0	0	0	0	0	0	0	0	0	0
I1	24/01/2007	0	0	0	0	0	0	0	8	16	0	0	0	0	39	0	0	5386	0
I1	15/02/2007																		
I1	15/03/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I1	19/04/2007	0	0	0	0	0	0	0	0	55	0	0	53	0	0	0	0	0	0
I1	17/05/2007	0	0	0	0	0	0	0	0	24	0	0	0	0	0	0	0	0	0

Lake	Date	Pediastrum duplex	Pediastrum simplex	Pteromonas sp.	Quadrigula sp.	Scenedesmus abundans	Scenedesmus arcuatus	Scenedesmus bicaudatus	Scenedesmus communis (2 cell)	Scenedesmus communis (4 cell)	Scenedesmus communis (8 cell)	Scenedesmus falcatus (2 cell)	Scenedesmus falcatus (4 cell)	Scenedesmus falcatus (8 cell)	Scenedesmus opoliensis (4 cell)	Scenedesmus opoliensis (2 cell)	Selenastrum sp.	Schroederia sp.	Staurastrum punctulatum
I1	14/06/2007	254	0	0	0	0	0	0	247	1486	220	0	0	0	0	0	0	0	0
I1	16/07/2007	0	0	0	0	0	0	0	18	18	0	0	0	0	0	0	0	0	0
I1	08/08/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I1	11/09/2007	0	0	0	0	0	0	0	1318	0	0	0	0	0	0	0	0	0	0
I1	15/10/2007	0	0	0	0	0	0	0	19	0	0	0	147	0	0	0	0	818	0
I1	15/11/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I1	13/12/2007	0	0	0	0	0	0	0	549	880	440	0	0	0	0	0	0	0	536
I1	17/01/2008	0	0	0	0	0	0	0	0	715	0	0	0	0	0	0	0	1527	0
I1	19/02/2008	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I1	18/03/2008	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1416	0
I2	22/03/2005	0	0	0	0	0	0	0	45	0	0	0	0	0	0	0	0	0	0
I2	15/04/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	11/05/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	09/06/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	06/07/2005	0	0	0	0	0	0	0	134	134	0	0	0	0	0	0	0	0	0
I2	01/08/2005	0	0	0	0	0	0	0	0	165	0	0	9627	0	0	0	0	0	0
I2	31/08/2005	0	0	0	0	0	0	359	0	440	0	0	0	0	0	0	0	0	0
I2	04/10/2005	0	0	0	0	0	0	0	0	0	0	0	63	0	0	0	0	0	0
I2	01/11/2005	0	0	0	0	0	0	0	13	0	0	0	0	0	0	0	0	0	0
I2	24/11/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	21/12/2005	0	0	0	0	0	0	0	55	0	0	0	0	0	0	0	0	0	0
I2	17/01/2006	0	0	0	0	0	0	0	0	55	0	0	107	0	0	0	0	5178	0
I2	15/02/2006	0	0	0	0	0	0	0	0	660	0	0	0	0	2171	0	0	2124	0
I2	15/03/2006	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	20/04/2006	0	0	0	0	0	0	0	0	1761	0	0	0	0	6078	1302	0	850	0
I2	18/05/2006	0	0	0	0	0	0	0	6027	16098	0	0	0	0	0	0	0	0	0
I2	15/06/2006	0	63	0	0	0	0	0	0	0	0	14	0	0	35	0	0	883	0
I2	13/07/2006	0	0	0	0	0	0	0	0	0	0	0	0	0	3545	0	0	0	0
I2	10/08/2006	0	0	0	0	0	0	1437	0	2817	0	342	0	0	1737	0	0	0	0
I2	07/09/2006	0	48	0	0	0	0	0	49	110	0	0	0	0	0	0	0	0	0
I2	04/10/2006	726	0	0	0	0	0	0	8	283	0	0	0	0	0	0	0	0	0
I2	02/11/2006	0	0	0	0	0	38	0	0	1224	102	50	0	0	252	0	0	0	0
I2	30/11/2006	0	0	0	0	0	0	0	0	376	0	0	0	0	0	0	0	22507	0
I2	20/12/2006	0	0	0	0	0	0	0	0	42	0	0	0	0	0	0	0	3674	0
I2	24/01/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	8677	0
I2	15/02/2007																		
I2	15/03/2007	0	0	0	0	0	0	0	0	49	0	0	0	0	0	0	0	5901	0
I2	19/04/2007	0	0	0	0	0	0	0	24	49	0	0	0	0	0	0	0	1180	0
I2	17/05/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	965	0	0	0	0
I2	14/06/2007	0	0	0	0	0	0	0	11	0	0	0	0	0	0	0	0	0	0
I2	16/07/2007	0	0	0	0	0	0	0	391	587	0	0	0	0	0	0	119	0	0
I2	08/08/2007	0	0	0	0	0	0	0	195	1174	0	190	0	0	1447	0	0	0	0
I2	11/09/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	15/10/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2862	0
I2	15/11/2007	0	0	0	0	0	0	0	0	220	0	0	0	0	0	0	0	14073	0
I2	13/12/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	28677	0
I2	17/01/2008	0	0	0	0	0	0	0	0	880	0	0	0	0	0	0	0	0	0
I2	19/02/2008	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	10408	0
I2	18/03/2008	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	22/03/2005	0	0	0	0	0	0	0	319	1278	0	0	0	0	0	0	0	0	0
I3	15/04/2005	0	0	0	0	0	0	0	0	75126	0	0	73023	0	0	0	0	0	0
I3	11/05/2005	0	0	0	0	0	0	0	0	988	0	0	240	0	0	0	0	0	0
I3	09/06/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	06/07/2005	10671	0	0	0	0	0	0	1648	2971	0	0	1284	0	13024	407	0	0	0
I3	01/08/2005	0	0	0	0	0	0	0	161	0	0	0	0	0	199	0	0	0	0
I3	31/08/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	124	0	0
I3	04/10/2005	0	0	0	0	0	0	0	1261	1404	0	0	0	0	0	0	0	0	0
I3	01/11/2005	0	0	0	0	0	0	0	0	4191	0	0	0	0	2583	0	0	0	0
I3	24/11/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	21/12/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	17/01/2006	0	0	0	0	0	0	0	0	39	0	0	0	0	0	0	0	0	0
I3	15/02/2006	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	15/03/2006	0	0	0	0	0	0	0	957	2876	0	0	0	0	0	0	0	0	0
I3	20/04/2006	0	0	0	0	0	0	0	0	689	0	0	0	0	0	0	0	0	0
I3	18/05/2006	0	0	0	0	0	0	0	0	28	0	0	0	0	0	0	0	0	0
I3	15/06/2006	0	0	0	0	0	0	0	0	37	0	0	0	0	0	0	0	0	0
I3	13/07/2006	0	0	0	0	0	0	0	0	805	0	0	0	0	0	0	0	0	0
I3	10/08/2006	0	0	0	0	0	0	0	879	1761	0	0	0	0	9768	0	0	0	0
I3	07/09/2006	0	0	0	0	0	0	0	7911	7043	0	0	0	0	8683	0	0	0	0
I3	04/10/2006	0	0	0	0	0	0	0	5742	5113	0	0	0	0	0	0	777	0	0
I3	02/11/2006	0	0	0	0	0	0	0	419	120	0	0	0	0	295	0	0	0	0
I3	30/11/2006	0	0	0	0	0	0	0	21	43	0	0	0	0	0	0	0	0	0
I3	20/12/2006	0	0	0	0	0	0	0	0	0	0	0	0	0	102	0	0	0	0

Lake	Date	Pediastrum duplex	Pediastrum simplex	Pteromonas sp.	Quadrigula sp.	Scenedesmus abundans	Scenedesmus arcuatus	Scenedesmus bicaudatus	Scenedesmus communis (2 cell)	Scenedesmus communis (4 cell)	Scenedesmus communis (8 cell)	Scenedesmus falcatus (2 cell)	Scenedesmus falcatus (4 cell)	Scenedesmus falcatus (8 cell)	Scenedesmus opcliensis (4 cell)	Scenedesmus opcliensis (2 cell)	Selenastrum sp.	Schroederia sp.	Staurastrum punctulatum
I3	24/01/2007	0	0	0	0	0	0	0	59	47	0	0	0	0	0	0	0	142	0
I3	15/02/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	15/03/2007	0	0	0	201	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	19/04/2007	0	0	0	0	0	0	0	1563	4695	0	0	0	0	1929	0	0	0	0
I3	17/05/2007	0	0	0	0	0	0	0	0	767	0	0	0	0	0	0	0	0	0
I3	14/06/2007	0	0	0	0	0	0	0	0	5175	0	0	0	0	0	0	0	0	0
I3	16/07/2007	0	0	0	0	0	0	0	439	880	0	0	0	0	0	0	0	0	0
I3	08/08/2007	0	0	6653	0	0	0	0	1266	7607	0	0	4929	19716	0	0	0	0	0
I3	11/09/2007	0	44652	0	0	0	0	0	7594	40568	0	4925	19716	0	0	0	0	0	0
I3	15/10/2007	0	0	0	0	0	0	0	1927	3509	0	0	1364	682	0	0	0	0	0
I3	15/11/2007	0	0	0	0	0	0	0	1209	660	0	0	1284	0	0	0	0	0	0
I3	13/12/2007	0	0	0	0	0	0	0	37	24	0	0	0	0	0	0	0	0	0
I3	17/01/2008	0	0	0	0	0	0	0	0	14	0	0	0	0	0	0	0	0	0
I3	19/02/2008	0	0	0	0	0	0	0	0	4695	0	0	0	0	0	0	0	0	0
I3	18/03/2008	0	0	0	0	0	0	0	0	16227	0	0	0	0	0	0	0	0	0

Lake	Date	Tetraedron sp.	Tetraedron caudatum	Tetraedron incus	Tetraedron minimum	Tetraedron regulare	Tetraedron triangulare	Tetradesmus sp.	Tetrastrum elegans	Ulothrix	Aphanizomenon flos-aquae	Anabaena flos-aquae	Anabaena chains	Anabaena aequalis	Aphonocapsa sp.	Chroococcus sp.	Oscillatoria spp.	Oscillatoria arghardii	Oscillatoria limentica	Merismopedia
C1	22/03/2005	0	0	0	0	0	0	0	3556	0	0	0	0	0	0	0	0	0	0	0
C1	15/04/2005	0	0	0	0	0	0	0	6135	0	10000	0	0	0	0	0	0	0	0	0
C1	11/05/2005	0	943	0	0	0	0	0	0	0	0	0	0	0	3756	0	0	0	0	0
C1	09/06/2005	0	31	0	0	0	0	0	0	0	0	165	0	0	0	0	0	0	0	0
C1	06/07/2005	0	157	0	0	0	0	0	157	0	0	0	0	0	0	0	0	0	0	0
C1	01/08/2005	0	21952	0	0	0	0	0	135227	0	0	0	0	0	0	0	0	0	0	0
C1	31/08/2005	0	22628	0	0	0	0	0	90151	0	0	0	0	0	0	0	0	0	0	43273
C1	04/10/2005	0	0	0	0	0	0	0	33117	0	0	0	0	0	0	0	178831	0	0	0
C1	01/11/2005	0	3620	0	0	0	0	0	9015	1551	0	0	0	0	0	0	0	0	0	0
C1	24/11/2005	0	75	0	0	0	0	0	751	0	0	460	0	0	0	0	45	0	0	0
C1	21/12/2005	0	0	0	0	0	40	0	559	0	0	0	0	0	0	0	0	0	0	0
C1	17/01/2006	0	0	0	0	0	78	0	313	0	0	0	0	0	0	0	0	0	0	0
C1	15/02/2006	0	0	0	0	0	0	0	7155	0	0	2576	0	0	0	0	0	0	0	0
C1	15/03/2006	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
C1	20/04/2006	0	0	0	0	0	2235	0	4470	0	0	0	0	0	0	0	0	0	0	0
C1	18/05/2006	0	0	0	0	0	0	0	59	0	0	0	0	0	0	0	0	0	0	0
C1	15/06/2006	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	9	0	0
C1	13/07/2006	0	0	0	0	0	0	0	3377	0	0	0	0	0	0	0	0	0	0	2594
C1	10/08/2006	0	0	0	0	0	30050	0	60101	0	0	0	0	0	0	0	0	1476879	0	47440
C1	07/09/2006	0	131	0	0	0	1100	0	88	0	0	0	0	0	0	0	0	401	0	34
C1	04/10/2006	0	0	0	0	0	2817	0	2817	0	0	0	0	0	0	0	0	0	7747	1623
C1	02/11/2006	0	38	0	0	0	117	0	117	0	0	0	0	0	0	0	47	0	0	50
C1	30/11/2006	0	8	0	0	0	0	0	8	0	0	0	0	0	0	0	0	0	0	0
C1	20/12/2006	0	0	0	0	0	2	0	19	0	0	0	0	0	0	0	0	0	0	0
C1	24/01/2007	0	0	0	0	0	0	0	45	0	0	32	0	0	0	0	0	0	0	0
C1	15/02/2007	0	0	0	0	0	0	0	15	0	0	819	0	0	0	0	0	301	0	36
C1	15/03/2007	0	0	0	0	0	0	0	15025	0	0	0	0	0	0	0	2254	0	0	0
C1	19/04/2007	0	0	0	0	0	0	0	9	0	0	0	0	0	0	0	0	0	0	0
C1	17/05/2007	0	0	0	0	0	0	0	0	0	1913	56	0	0	0	0	0	0	0	0
C1	14/06/2007	0	0	0	0	0	0	0	40568	0	0	128520	0	0	0	0	4868	0	0	0
C1	16/07/2007	0	0	0	0	0	16558	0	20698	0	0	0	0	0	0	0	0	0	0	2119
C1	08/08/2007	0	0	0	0	0	24341	0	27045	0	0	0	0	0	0	0	0	0	1352	692
C1	11/09/2007	0	0	0	0	0	16558	0	33117	0	0	0	0	0	0	0	0	163597	0	0
C1	15/10/2007	0	1414	0	0	0	0	0	2817	0	0	0	0	0	0	0	0	191628	2817	0
C1	15/11/2007	0	471	0	0	0	470	0	939	0	0	0	0	0	0	0	0	78507	704	0
C1	13/12/2007																			
C1	17/01/2008	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	42	0	0
C1	19/02/2008	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
C1	18/03/2008	0	0	0	0	0	0	0	0	0	0	24341	0	0	0	0	27045	0	0	0
C2	22/03/2005	0	806	0	0	0	0	0	17386	0	0	0	0	0	0	0	0	0	414	0
C2	15/04/2005	0	0	0	0	0	3371	0	4495	0	0	0	0	0	0	0	0	0	0	0
C2	11/05/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
C2	09/06/2005	0	0	0	0	0	0	0	1534	0	0	46933	0	0	0	0	920	0	0	0
C2	06/07/2005	2037	0	0	0	0	0	0	2028	0	0	0	0	0	0	0	1217	0	0	0
C2	01/08/2005	53858	0	0	0	0	0	0	80466	0	0	0	0	0	0	0	0	0	0	51498
C2	31/08/2005	0	0	0	0	7903	0	0	64909	0	0	0	0	0	0	0	68154	0	0	0
C2	04/10/2005	0	0	0	0	0	150252	0	60101	0	0	0	0	0	0	0	0	22838	345580	0
C2	01/11/2005	0	0	0	0	0	0	0	1998	0	0	0	0	0	0	0	0	0	0	256
C2	24/11/2005	577	0	0	0	0	0	0	575	0	0	0	0	0	0	0	0	0	0	0
C2	21/12/2005	689	0	0	0	0	0	0	0	0	0	0	0	0	0	0	37	0	0	0
C2	17/01/2006	0	0	0	0	0	42	0	296	0	0	0	0	0	0	0	0	0	0	0
C2	15/02/2006	0	0	0	0	0	0	0	2633	0	0	0	0	0	0	0	0	0	0	0
C2	15/03/2006	707	0	0	0	0	0	0	3874	0	0	380	0	0	0	0	211	0	0	0
C2	20/04/2006	2244	0	0	0	0	0	0	4470	0	0	0	0	0	0	0	0	0	0	0
C2	18/05/2006	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
C2	15/06/2006	0	0	0	0	0	0	0	0	0	0	0	0	0	98	0	88	0	0	0
C2	13/07/2006	38561	0	0	0	0	0	0	0	0	0	0	0	0	0	0	11522	0	0	83575
C2	10/08/2006	130337	0	0	0	0	0	0	32454	0	0	0	0	0	0	0	3816647	0	0	436188
C2	07/09/2006	48876	0	0	0	0	0	0	24341	0	0	0	0	0	0	0	34077	0	0	17655
C2	04/10/2006	10861	0	0	0	0	0	0	3606	0	0	0	0	0	0	0	10818	0	0	18001
C2	02/11/2006	140	0	0	0	0	0	0	349	0	0	0	0	0	0	0	126	0	0	0
C2	30/11/2006	0	0	0	0	0	0	0	26	0	0	0	0	0	0	0	31	0	0	3
C2	20/12/2006	12	0	0	0	0	0	0	25	0	0	362	0	0	0	0	15	0	0	0
C2	24/01/2007	0	0	0	0	0	0	0	0	0	0	117	0	0	0	0	13	0	0	0
C2	15/02/2007	0	0	0	0	0	57	0	85	0	0	0	0	0	0	0	17	0	0	15
C2	15/03/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	337	0	0	0
C2	19/04/2007	0	69	0	0	0	634	0	282	0	0	0	0	0	0	0	0	54	0	0
C2	17/05/2007	0	0	0	0	0	0	0	100	0	0	1552	0	0	0	0	0	0	0	0
C2	14/06/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
C2	16/07/2007	0	0	0	0	0	1202	0	3606	0	0	0	0	0	0	0	0	914	0	0
C2	08/08/2007	0	988	0	0	3457	0	0	507	436	0	0	0	0	0	0	304	0	0	65
C2	11/09/2007	0	12348	0	0	0	12678	0	25355	0	0	27383	0	0	0	0	0	9635	0	0
C2	15/10/2007	0	0	0	0	0	2817	0	2817	0	0	12678	0	0	0	0	0	0	125367	0

Lake	Date	Tetraedron sp.	Tetraedron caudatum	Tetraedron incus	Tetraedron minimum	Tetraedron regulare	Tetraedron triangulare	Tetradesmus sp.	Tetrastrum elegans	Ulothrix	Aphanizomenon flos-aquae	Anabaena flos-aquae	Anabaena chains	Anabaena aequalis	Aphonocapsa sp.	Chroococcus sp.	Oscillatoria spp.	Oscillatoria arghardii	Oscillatoria limentica	Merismopedia
C2	15/11/2007	0	0	0	0	0	528	0	352	0	0	0	0	0	0	0	0	26630	264	0
C2	13/12/2007	0	0	0	0	0	0	0	11	0	0	0	0	0	0	0	0	376	0	0
C2	17/01/2008	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5	0	0	0
C2	19/02/2008	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	714	0	0
C2	18/03/2008	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
C3	22/03/2005	0	0	0	0	0	15234	0	0	0	0	0	0	0	0	0	0	0	0	509
C3	15/04/2005	10172	0	0	0	0	20263	0	0	0	33029	0	0	0	0	0	0	0	0	0
C3	11/05/2005	108	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
C3	09/06/2005	0	0	0	0	0	0	0	0	0	0	4673	0	0	0	0	19473	0	0	0
C3	06/07/2005	332	0	0	0	0	0	0	0	0	0	0	0	0	0	0	99	0	0	0
C3	01/08/2005	5657	16464	0	0	0	0	0	90151	0	0	0	0	0	0	0	0	4282	0	56254
C3	31/08/2005	0	258046	0	0	322558	993504	0	0	0	215922	0	0	0	0	0	874284	0	0	839313
C3	04/10/2005	22166	0	0	0	0	36796	0	0	0	0	0	0	0	0	0	211948	0	0	4710
C3	01/11/2005	1609	0	0	0	0	801	0	0	0	0	0	0	0	0	0	481	0	0	205
C3	24/11/2005	0	0	0	0	231	711	0	0	0	0	0	0	0	0	0	0	0	0	0
C3	21/12/2005	94	0	0	0	0	0	0	1583	0	0	0	0	0	0	0	0	0	0	0
C3	17/01/2006	0	0	0	0	0	0	0	2641	0	0	0	0	0	0	0	0	0	0	0
C3	15/02/2006	0	0	0	0	0	0	0	94619	0	0	0	0	0	0	0	0	0	0	0
C3	15/03/2006	0	0	0	0	0	0	0	9860	0	0	0	0	0	0	0	0	0	0	0
C3	20/04/2006	0	4032	0	0	0	8279	0	8279	0	0	0	0	0	0	0	0	0	0	0
C3	18/05/2006	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
C3	15/06/2006	0	51	0	0	0	0	0	52	0	3912	3381	0	0	0	0	0	0	0	0
C3	13/07/2006	0	16128	0	0	0	33117	0	66234	0	0	226519	0	0	0	0	0	25169	0	186514
C3	10/08/2006	0	21876	0	0	0	11230	0	44920	0	0	0	0	0	0	0	0	1689879	0	206990
C3	07/09/2006	0	0	0	0	0	0	0	66234	0	0	0	0	0	0	0	0	742479	0	150483
C3	04/10/2006	5637	0	0	0	0	0	2190	8422	0	0	0	0	0	0	0	0	21337	0	0
C3	02/11/2006	0	980	0	0	0	201	0	2012	0	0	0	0	0	0	0	0	306	0	824
C3	30/11/2006	0	0	0	0	0	0	0	176	0	0	0	0	0	0	0	0	33	0	0
C3	20/12/2006	0	0	0	0	0	0	0	77	0	0	40	0	0	0	0	0	0	0	23
C3	24/01/2007	0	7	0	0	0	7	0	14	0	0	10	0	0	0	0	0	16	0	0
C3	15/02/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	45	0	23
C3	15/03/2007	0	0	0	0	0	0	0	4808	0	0	0	0	0	0	0	0	1370	0	0
C3	19/04/2007	0	0	0	0	0	3345	0	0	0	0	0	314	0	0	0	0	0	0	0
C3	17/05/2007	0	0	0	0	0	0	0	176	0	0	0	941	0	0	0	0	0	176	0
C3	14/06/2007	0	0	0	0	0	0	0	9015	0	0	0	8032	0	0	0	0	0	0	0
C3	16/07/2007	603	585	0	0	0	601	0	3606	0	0	0	0	0	0	0	0	0	0	539
C3	08/08/2007	0	0	0	0	0	157	0	313	0	0	0	0	0	0	0	0	714	0	0
C3	11/09/2007	0	2734	0	0	0	2807	0	11230	0	0	0	0	0	0	0	0	25604	0	719
C3	15/10/2007	0	686	0	0	0	1409	0	704	0	0	0	0	0	0	0	0	137030	0	0
C3	15/11/2007	0	2688	0	0	0	2760	0	5519	0	0	0	0	0	0	0	0	326145	4140	0
C3	13/12/2007	0	0	0	0	0	0	0	34	0	0	0	0	0	0	0	0	411	6	0
C3	17/01/2008	0	0	0	0	0	75	0	56	0	0	115	0	0	0	0	0	71	0	0
C3	19/02/2008	0	0	0	0	0	0	0	3756	0	0	0	0	0	0	0	0	5710	0	0
C3	18/03/2008	0	0	0	0	0	0	0	25355	0	0	0	0	0	0	0	7607	0	0	0
I1	22/03/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I1	15/04/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I1	11/05/2005	0	0	0	0	0	0	0	0	0	0	320	0	0	0	0	0	0	0	0
I1	09/06/2005	0	0	0	0	0	12	0	0	0	0	72	0	0	0	0	30	0	0	0
I1	06/07/2005	0	0	0	0	0	117	0	0	0	34444	3317	0	0	0	117	0	0	0	0
I1	01/08/2005	0	0	0	0	0	176	0	0	0	0	0	0	0	0	0	70	0	0	0
I1	31/08/2005	0	0	0	0	0	138	0	13	22	0	1444	0	0	0	0	0	0	0	0
I1	04/10/2005	0	0	0	0	0	0	0	0	0	0	254	0	0	0	0	0	0	0	0
I1	01/11/2005	0	0	0	0	0	0	0	6	0	0	23	0	0	0	0	7	0	0	3
I1	24/11/2005	0	0	0	0	0	0	0	0	0	0	16	0	0	0	0	14	0	0	0
I1	21/12/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	8	0	0	0
I1	17/01/2006	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I1	15/02/2006	0	0	0	0	0	0	0	230	0	0	0	0	0	0	0	0	0	0	0
I1	15/03/2006	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I1	20/04/2006	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	42	0	0	0
I1	18/05/2006	0	0	0	0	0	0	0	0	0	42258	0	0	0	0	0	0	0	0	0
I1	15/06/2006	0	0	0	0	0	0	0	0	0	21260	1690	0	0	0	0	0	0	0	0
I1	13/07/2006	0	0	0	0	0	0	0	0	0	62604	3457	0	0	0	0	7201	0	0	0
I1	10/08/2006	0	0	0	0	0	0	0	0	0	0	61866	0	0	0	0	0	0	0	0
I1	07/09/2006	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	423	0	0	0
I1	04/10/2006	354	0	0	0	0	0	0	0	151	0	0	0	0	0	0	106	0	0	0
I1	02/11/2006	0	0	0	0	0	0	0	1053	0	0	505	0	0	0	0	15371	0	0	0
I1	30/11/2006	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	55	0	0	0
I1	20/12/2006	0	0	0	0	0	0	0	0	5	0	0	0	0	0	0	0	0	0	0
I1	24/01/2007	0	0	0	0	0	13	0	0	0	0	0	0	0	0	0	0	0	0	0
I1	15/02/2007																			
I1	15/03/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I1	19/04/2007	22	0	0	0	0	22	0	0	0	0	0	0	0	0	0	0	0	55	51
I1	17/05/2007	0	0	0	0	0	0	0	57	0	0	177	0	0	0	0	0	0	0	0

Lake	Date	Tetraedron sp.	Tetraedron caudatum	Tetraedron incus	Tetraedron minimum	Tetraedron regulare	Tetraedron triangulare	Tetradesmus sp.	Tetrastrum elegans	Ulothrix	Aphanizomenon flos-aquae	Anabaena flos-aquae	Anabaena chains	Anabaena aequalis	Aphonocapsa sp.	Chroococcus sp.	Oscillatoria spp.	Oscillatoria arghardii	Oscillatoria limentica	Merismopedia
I1	14/06/2007	0	0	0	0	0	0	0	88	0	0	285	0	0	0	0	0	33	0	0
I1	16/07/2007	0	0	0	0	0	0	0	0	25	0	0	0	0	0	0	9	0	0	0
I1	08/08/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I1	11/09/2007	0	0	0	0	0	0	0	0	0	0	1078	0	0	0	0	0	268	0	0
I1	15/10/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	160	0	0
I1	15/11/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I1	13/12/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	134	0	0
I1	17/01/2008	0	0	0	0	0	132	0	0	0	0	143	0	0	0	0	0	0	0	0
I1	19/02/2008	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I1	18/03/2008	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	22/03/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	15/04/2005	0	0	0	0	0	0	0	51	0	0	0	0	0	0	0	0	0	0	0
I2	11/05/2005	0	0	0	0	0	0	0	0	0	0	331	0	0	0	0	0	0	0	0
I2	09/06/2005	0	0	0	0	0	0	0	0	0	4099	2414	0	0	0	0	0	0	0	0
I2	06/07/2005	0	0	0	0	0	0	0	0	0	2098	463	0	0	0	0	0	0	0	0
I2	01/08/2005	0	0	0	0	129	0	0	0	0	0	2678	0	0	0	0	0	0	0	0
I2	31/08/2005	0	354	0	0	0	0	0	0	0	0	0	0	0	1409	0	0	0	0	0
I2	04/10/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	20	0	23
I2	01/11/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	24/11/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	21/12/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	17/01/2006	0	0	0	0	0	0	0	0	38	0	0	0	0	0	0	0	0	88	0
I2	15/02/2006	0	0	0	0	0	176	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	15/03/2006	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	20/04/2006	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	18/05/2006	0	0	0	0	0	0	0	12879	0	0	0	0	0	0	0	0	0	0	0
I2	15/06/2006	0	0	0	0	0	0	0	0	0	636	70	0	0	0	0	55	0	0	0
I2	13/07/2006	0	0	0	0	0	1150	0	0	0	0	0	199521	0	0	0	0	0	0	0
I2	10/08/2006	0	0	282	0	0	282	0	0	0	0	0	1004	0	0	0	0	5139	35920	0
I2	07/09/2006	0	0	0	0	0	18	0	0	0	0	0	0	0	0	0	0	67	13	0
I2	04/10/2006	0	0	0	0	0	0	0	0	0	0	18	0	0	0	0	0	0	0	0
I2	02/11/2006	0	0	0	0	0	0	0	41	0	0	529	0	0	0	0	0	0	0	0
I2	30/11/2006	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	20/12/2006	0	0	0	0	0	0	0	0	0	0	312	0	0	0	0	0	0	0	0
I2	24/01/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	15/02/2007																			
I2	15/03/2007	0	38	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	19/04/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	157	0	0	0	0
I2	17/05/2007	0	0	0	0	0	313	0	313	0	0	1183	0	0	0	0	94	0	0	0
I2	14/06/2007	0	0	0	0	0	0	0	0	0	0	106	0	0	0	0	63	0	0	0
I2	16/07/2007	0	305	0	0	0	470	0	157	0	17858	0	0	0	0	0	0	0	0	0
I2	08/08/2007	0	0	0	0	0	157	0	0	0	0	282	0	0	0	0	0	0	0	0
I2	11/09/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	30	0	20
I2	15/10/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	15/11/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	13/12/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2434
I2	17/01/2008	0	0	0	0	0	0	0	0	0	0	127	0	0	0	0	0	0	0	0
I2	19/02/2008	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I2	18/03/2008	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	22/03/2005	0	0	0	0	0	0	0	1023	0	0	0	0	0	0	0	0	0	0	0
I3	15/04/2005	0	29269	0	0	0	30050	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	11/05/2005	0	0	0	99	0	0	0	0	0	0	0	0	0	0	0	59	0	0	0
I3	09/06/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	06/07/2005	0	0	0	0	0	0	0	792	0	0	570	0	0	0	0	0	201	0	34
I3	01/08/2005	0	0	0	0	0	259	0	129	0	0	23	0	0	0	0	39	0	0	0
I3	31/08/2005	328	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	04/10/2005	0	219	0	0	0	0	0	449	0	0	0	0	0	0	0	0	0	0	0
I3	01/11/2005	0	0	0	0	0	0	0	838	0	0	0	0	0	0	0	0	0	0	0
I3	24/11/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	21/12/2005	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
I3	17/01/2006	0	0	0	0	0	0	0	0	0	0	383	0	0	0	0	0	0	0	0
I3	15/02/2006	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	15/03/2006	0	0	0	0	0	2301	0	0	0	0	828	0	0	0	0	16565	0	0	0
I3	20/04/2006	184	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	18/05/2006	22	0	0	0	0	0	0	44	0	0	16	0	0	0	0	26	0	0	0
I3	15/06/2006	0	0	0	0	0	0	0	59	0	0	380	0	0	0	0	0	0	0	0
I3	13/07/2006	0	0	0	0	0	0	0	322	138	0	637	0	0	0	0	193	0	0	0
I3	10/08/2006	0	2401	0	704	0	1761	0	0	0	0	0	0	2183	0	0	0	0	0	0
I3	07/09/2006	0	2744	0	1409	0	7043	0	11269	0	0	0	0	0	0	0	845	0	0	0
I3	04/10/2006	0	0	0	511	0	2556	0	1534	0	0	0	0	0	0	0	0	0	0	524
I3	02/11/2006	0	0	0	0	0	96	0	0	0	0	863	0	0	0	0	0	0	0	0
I3	30/11/2006	0	0	0	0	0	17	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	20/12/2006	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Lake	Date	Tetraedron sp.	Tetraedron caudatum	Tetraedron incus	Tetraedron minimum	Tetraedron regulare	Tetraedron triangulare	Tetradesmus sp.	Tetrastrum elegans	Ulothrix	Aphanizomenon flos-aquae	Anabaena flos-aquae	Anabaena chains	Anabaena aequalis	Aphonocapsa sp.	Chroococcus sp.	Oscillatoria spp.	Oscillatoria arghardii	Oscillatoria limentica	Merismopedia
I3	24/01/2007	0	18	0	0	0	0	0	19	0	0	0	0	0	0	0	0	0	0	0
I3	15/02/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	15/03/2007	0	187	0	0	0	0	0	192	0	0	0	0	0	0	0	0	0	0	0
I3	19/04/2007	0	0	0	0	0	626	0	1878	0	0	0	0	0	0	0	0	0	0	0
I3	17/05/2007	0	0	0	0	0	0	0	0	0	0	1640	1640	0	0	0	0	0	0	0
I3	14/06/2007	0	4032	0	0	0	0	0	4140	0	0	0	0	0	0	0	0	3146	0	0
I3	16/07/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	496
I3	08/08/2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	11/09/2007	0	3951	0	0	0	8114	0	4057	0	0	0	0	0	0	0	0	6166	0	0
I3	15/10/2007	0	0	0	0	0	140	0	983	0	0	0	0	0	0	0	0	107	0	0
I3	15/11/2007	0	0	0	0	0	880	0	528	0	0	0	0	0	0	0	0	0	0	406
I3	13/12/2007	0	19	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	17/01/2008	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	19/02/2008	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I3	18/03/2008	0	0	0	0	0	0	0	6491	0	0	0	0	0	0	0	0	0	0	0

Lake	Date	Microcystis sp.	Oscillatoria redekei	Spirulina	Cryptomonas spp.	Rhodomonas spp.	Euglena sp.	Euglena acus	Phacus sp.	Phacus caudatum	Trachelomonas sp.	Trachelomonas bacillifera	Ceratium hirundinella	Gymnodinium sp.	Peridinium sp.	Dinobryon cylindricum var. alpinum	Mallomonas sp.
C1	22/03/2005	0	0	0	8172	23107	0	0	0	0	0	0	0	0	29633	0	0
C1	15/04/2005	0	0	0	126885	152818	0	0	0	0	0	0	0	0	76688	0	0
C1	11/05/2005	0	0	0	110058	3729	2817	0	0	0	0	0	0	918	0	0	0
C1	09/06/2005	0	0	0	985	287	0	0	0	0	0	0	0	0	0	0	0
C1	06/07/2005	0	0	0	68696	4181	0	0	0	0	0	0	0	0	0	0	2334
C1	01/08/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
C1	31/08/2005	3155297	0	0	103584	16272	0	0	0	0	0	0	0	0	0	0	0
C1	04/10/2005	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	98754
C1	01/11/2005	0	0	0	24860	10414	0	0	0	0	0	0	0	31740	22538	0	16130
C1	24/11/2005	0	0	0	9664	9651	0	0	0	0	0	0	0	0	3755	0	672
C1	21/12/2005	0	0	0	8998	3866	0	0	0	0	235	0	0	0	2997	0	0
C1	17/01/2006	0	0	0	809	961	0	0	0	0	0	0	0	0	0	0	0
C1	15/02/2006	0	0	0	177572	10332	0	0	0	0	25242	0	0	0	0	0	0
C1	15/03/2006	0	0	0	57814	12884	585	0	0	0	0	0	0	4578	25597	0	6979
C1	20/04/2006	0	0	0	82182	0	0	0	2494	0	0	0	0	0	55879	0	13330
C1	18/05/2006	0	0	0	28728	339	1174	0	0	0	115	0	0	0	0	0	0
C1	15/06/2006	0	0	0	1432	467	144	0	0	0	0	0	0	0	0	0	0
C1	13/07/2006	0	0	0	31043	0	0	0	0	0	6619	0	0	0	42214	0	0
C1	10/08/2006	841412	0	0	69056	0	10017	0	0	0	39266	0	0	19593	0	0	0
C1	07/09/2006	0	0	0	3945	636	44	0	0	0	0	0	0	86	0	0	0
C1	04/10/2006	0	0	0	6474	2034	0	0	0	0	0	0	0	0	0	0	8401
C1	02/11/2006	0	0	0	3147	141	39	0	0	0	0	0	0	0	0	0	0
C1	30/11/2006	0	0	0	3201	136	0	0	0	0	0	0	0	0	0	0	0
C1	20/12/2006	0	0	0	363	106	0	0	0	0	14	0	0	0	0	0	21
C1	24/01/2007	0	0	0	596	203	0	0	0	0	110	0	0	0	141	0	134
C1	15/02/2007	0	0	0	742	64	0	0	0	0	58	0	0	0	0	0	0
C1	15/03/2007	0	0	0	207168	100346	0	0	0	0	22087	0	0	0	93908	0	0
C1	19/04/2007	0	0	0	734	129	0	0	0	0	0	0	0	0	0	0	0
C1	17/05/2007	0	0	0	3776	735	0	0	0	0	0	0	0	0	0	0	0
C1	14/06/2007	0	0	0	503418	41006	0	0	0	0	0	0	0	0	0	0	48390
C1	16/07/2007	347727	0	0	28538	62765	0	0	0	0	0	0	0	0	0	0	0
C1	08/08/2007	151454	0	0	37290	21479	0	0	0	0	10602	0	0	5290	0	0	0
C1	11/09/2007	927271	0	0	190256	0	0	0	0	0	0	0	0	0	0	0	0
C1	15/10/2007	0	0	0	297804	14238	0	0	0	0	0	0	0	0	0	0	4200
C1	15/11/2007	0	0	0	20501	4407	0	0	0	0	7362	0	0	0	0	0	0
C1	13/12/2007																
C1	17/01/2008	0	0	0	898	222	11	0	80	0	32	0	0	0	69	0	0
C1	19/02/2008	0	0	0	302120	62377	0	0	0	0	0	0	0	0	0	0	0
C1	18/03/2008	0	0	0	51792	16272	0	0	0	0	0	0	0	88168	0	0	403247
C2	22/03/2005	0	0	0	11415	14944	0	0	0	0	0	0	0	32388	10349	0	4938
C2	15/04/2005	0	0	0	142033	53550	0	0	0	0	0	0	0	6594	28094	0	0
C2	11/05/2005	0	0	0	10222	947	0	0	0	0	2937	0	0	0	0	0	0
C2	09/06/2005	0	0	0	52869	21040	0	0	0	0	0	0	0	0	0	0	0
C2	06/07/2005	0	0	0	629272	49793	0	0	0	0	0	0	0	55546	0	0	0
C2	01/08/2005	0	0	0	924550	367943	0	0	0	0	0	0	0	157391	0	0	0
C2	31/08/2005	227181	0	0	242386	82012	0	0	0	0	0	0	0	0	0	0	0
C2	04/10/2005	841412	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
C2	01/11/2005	0	0	0	0	6491	499	0	0	0	0	0	0	0	18730	0	1489
C2	24/11/2005	0	0	0	14539	25193	0	0	0	0	0	0	0	0	9586	0	0
C2	21/12/2005	0	0	0	4444	4910	0	0	0	0	0	0	0	122	1560	0	0
C2	17/01/2006	0	0	0	4951	1525	0	0	0	0	497	0	0	0	528	0	504
C2	15/02/2006	0	0	0	8775	0	0	0	0	0	0	0	0	773	0	0	0
C2	15/03/2006	0	0	0	46127	9407	352	0	0	0	0	0	0	689	0	0	0
C2	20/04/2006	0	0	0	210592	187199	0	0	0	0	0	0	0	0	55879	0	13330
C2	18/05/2006	0	0	0	20071	308	639	0	0	0	0	0	0	0	0	0	0
C2	15/06/2006	0	0	0	3208	1114	294	0	0	0	0	0	0	48	0	0	0
C2	13/07/2006	0	0	0	397174	27730	0	0	0	0	0	0	0	37563	0	0	0
C2	10/08/2006	908725	0	0	223741	538939	0	0	0	0	0	0	0	0	405681	0	0
C2	07/09/2006	227181	0	0	18645	17574	0	0	0	0	0	0	0	0	0	0	0
C2	04/10/2006	0	0	0	74580	46864	0	0	0	0	0	0	0	0	0	0	0
C2	02/11/2006	0	0	0	13483	1664	0	0	0	0	0	0	0	0	0	0	625
C2	30/11/2006	0	0	0	16904	188	0	0	0	0	665	0	0	0	0	0	0
C2	20/12/2006	0	0	0	1043	301	12	0	0	0	0	0	0	0	0	0	0
C2	24/01/2007	0	0	0	347	50	0	0	0	0	105	0	0	0	0	0	28
C2	15/02/2007	0	0	0	1958	984	0	0	0	0	167	0	0	0	355	0	0
C2	15/03/2007	0	0	0	47775	33672	0	0	0	0	8810	0	0	0	14047	0	1676
C2	19/04/2007	0	0	0	14081	3356	0	0	0	0	0	0	0	0	0	0	210
C2	17/05/2007	0	0	0	21428	3982	0	0	112	44	0	0	0	0	0	0	598
C2	14/06/2007	0	0	0	228307	23910	0	0	0	0	0	0	0	0	0	0	0
C2	16/07/2007	0	0	0	22098	32111	0	0	0	0	0	0	0	0	0	0	0
C2	08/08/2007	0	0	0	25637	22700	0	0	0	0	0	0	0	0	0	0	12097
C2	11/09/2007	0	0	0	436994	82379	0	0	0	0	0	0	0	0	158469	0	0
C2	15/10/2007	78882	0	0	369018	16272	0	0	0	0	2761	0	0	5511	0	0	0

Lake	Date	Microcystis sp.	Oscillatoria redekei	Spirulina	Cryptomonas spp.	Rhodomonas spp.	Euglena sp.	Euglena acus	Phacus sp.	Phacus caudatum	Trachelomonas sp.	Trachelomonas bacillifera	Ceratium hirundinella	Gymnodinium sp.	Peridinium sp.	Dinobryon cylindricum var. apinum	Mallomonas sp.
C2	15/11/2007	9860	0	0	8497	1398	0	0	0	0	690	0	0	0	0	0	0
C2	13/12/2007	0	0	0	936	318	11	0	123	0	0	0	0	65	0	0	66
C2	17/01/2008	0	0	0	380	132	0	0	0	0	0	0	0	9	0	0	0
C2	19/02/2008	0	0	0	53950	12882	0	0	0	1660	0	0	0	0	0	0	0
C2	18/03/2008	0	0	0	0	3905	0	0	0	0	0	0	0	21160	0	0	5377
C3	22/03/2005	37091	0	0	10654	478	0	0	0	0	0	0	0	0	0	0	0
C3	15/04/2005	0	0	0	124171	195065	0	0	0	0	0	0	0	19817	0	0	0
C3	11/05/2005	0	0	0	4927	155	0	0	0	0	0	0	0	0	0	0	0
C3	09/06/2005	0	0	0	522063	32805	0	0	0	0	0	0	0	0	0	0	0
C3	06/07/2005	0	0	0	33485	837	0	0	0	0	0	0	0	1619	0	0	0
C3	01/08/2005	0	0	0	233064	475965	0	0	0	0	0	0	0	33063	0	0	16802
C3	31/08/2005	0	0	0	3805122	2869241	0	0	0	0	0	0	0	647765	0	0	0
C3	04/10/2005	0	0	0	152205	69074	7359	0	0	0	0	0	0	14395	0	0	0
C3	01/11/2005	0	1447	0	12890	4918	401	0	0	0	0	0	0	0	0	0	0
C3	24/11/2005	0	122	0	23427	13523	0	0	0	0	0	0	0	1391	2964	0	0
C3	21/12/2005	2608	0	0	4922	3967	0	0	0	0	0	0	0	273	0	0	0
C3	17/01/2006	0	0	0	15780	5212	0	0	0	0	0	0	0	0	4402	0	0
C3	15/02/2006	0	0	0	331589	30742	0	0	0	0	0	0	0	9254	88706	0	0
C3	15/03/2006	0	0	0	71214	14916	0	0	0	0	1841	0	0	3674	5869	0	8401
C3	20/04/2006	0	0	0	190256	23910	0	0	0	0	0	0	0	0	0	0	37033
C3	18/05/2006	0	0	0	6474	56	322	0	0	0	76	0	0	0	0	0	0
C3	15/06/2006	0	0	0	2877	829	0	0	0	0	19735	0	0	0	0	0	0
C3	13/07/2006	0	0	0	989332	573848	0	0	0	0	259636	0	0	32388	827920	0	0
C3	10/08/2006	0	0	0	258064	218916	0	0	0	0	0	0	0	0	0	0	0
C3	07/09/2006	927271	0	0	494666	239103	0	0	0	0	97363	0	0	0	0	0	0
C3	04/10/2006	0	0	0	116129	34459	0	0	0	0	0	0	0	0	0	0	0
C3	02/11/2006	0	0	0	59653	12785	201	0	0	0	1972	0	0	394	0	0	0
C3	30/11/2006	0	0	0	4855	795	0	0	0	0	345	0	0	344	0	0	0
C3	20/12/2006	0	0	0	1366	413	0	0	0	0	86	0	0	0	0	0	0
C3	24/01/2007	0	0	0	1448	124	0	0	0	0	0	0	0	0	90	0	0
C3	15/02/2007	0	0	0	2832	1610	0	0	0	0	115	0	0	0	0	0	0
C3	15/03/2007	0	0	0	80105	46430	601	0	1341	0	0	0	0	0	0	0	1792
C3	19/04/2007	0	0	0	29538	9280	0	0	0	0	0	0	0	0	2201	0	525
C3	17/05/2007	0	0	0	50578	7119	0	0	590	0	690	0	0	0	2201	0	0
C3	14/06/2007	0	0	0	528278	9763	0	0	0	0	0	0	0	0	112689	0	80649
C3	16/07/2007	50485	0	0	23479	22130	0	0	2012	0	0	0	0	0	7513	0	0
C3	08/08/2007	8765	0	0	5395	5763	0	0	0	0	0	0	0	0	5869	0	0
C3	11/09/2007	235828	0	0	245160	6081	0	0	0	0	11005	0	0	10983	0	0	0
C3	15/10/2007	39441	0	0	105202	11696	0	0	0	0	4141	0	0	0	0	0	0
C3	15/11/2007	0	0	0	107812	11955	0	0	0	0	0	0	0	2699	0	0	0
C3	13/12/2007	0	0	0	906	171	0	16	314	0	0	0	0	0	0	0	0
C3	17/01/2008	0	0	0	3108	461	0	0	0	0	221	0	0	0	0	0	0
C3	19/02/2008	0	0	0	319384	78650	0	0	0	0	14725	0	0	14695	46954	0	0
C3	18/03/2008	0	0	0	262197	54919	0	0	14148	0	0	0	0	0	0	0	0
I1	22/03/2005	0	0	0	11021	5828	0	0	0	0	229	0	0	0	0	0	2093
I1	15/04/2005	0	0	0	5067	297	27	0	0	0	376	0	0	0	0	0	0
I1	11/05/2005	0	0	0	8552	1602	0	0	0	0	201	0	0	67	427	0	0
I1	09/06/2005	699	0	0	2238	965	100	0	0	0	0	0	0	98	0	0	0
I1	06/07/2005	0	0	0	10520	1271	293	0	0	0	4256	0	0	0	8070	0	0
I1	01/08/2005	4930	0	0	2023	763	0	0	0	0	0	0	0	0	0	0	0
I1	31/08/2005	0	0	0	2139	390	25	0	0	0	74	0	0	0	1100	0	38
I1	04/10/2005	0	0	0	6127	8863	101	0	0	0	0	0	0	0	0	0	0
I1	01/11/2005	0	0	0	1255	436	29	0	0	0	0	0	0	0	0	0	17
I1	24/11/2005	0	0	0	1994	106	0	0	0	0	0	0	0	0	0	0	0
I1	21/12/2005	0	0	0	3960	64	25	0	0	0	0	0	0	0	0	0	0
I1	17/01/2006	0	0	0	1178	65	0	0	0	0	54	0	0	0	0	0	0
I1	15/02/2006	0	0	0	9340	1606	0	0	0	0	2856	0	0	0	0	0	0
I1	15/03/2006	0	0	0	33924	23670	0	0	0	85	1879	0	0	0	0	0	0
I1	20/04/2006	0	0	0	1295	7119	0	0	0	0	138	0	0	0	0	0	0
I1	18/05/2006	0	0	0	24601	36409	1409	0	0	0	0	0	0	0	0	0	0
I1	15/06/2006	0	0	0	1559	377	52	0	0	0	0	0	0	0	0	0	0
I1	13/07/2006	604921	0	0	176522	2600	1200	0	1339	0	0	0	0	0	0	0	0
I1	10/08/2006	157765	0	0	17803	509	0	0	1572	0	0	0	0	0	0	0	0
I1	07/09/2006	828265	0	0	247630	3051	0	0	786	0	0	0	0	0	88038	0	0
I1	04/10/2006	103533	0	0	92659	127	0	0	0	0	0	0	0	0	2201	0	0
I1	02/11/2006	9826	0	0	268547	0	0	0	0	0	0	0	0	0	0	0	0
I1	30/11/2006	0	0	0	30597	232	0	0	0	0	0	0	0	0	0	0	0
I1	20/12/2006	501	0	0	3520	26	0	0	0	0	0	0	0	0	0	0	0
I1	24/01/2007	0	0	0	0	91	13	0	0	0	0	0	0	0	0	0	0
I1	15/02/2007																
I1	15/03/2007	0	0	0	213932	5313	0	0	0	0	5048	0	0	0	0	0	0
I1	19/04/2007	0	0	0	7384	286	0	0	0	0	475	0	0	0	0	0	0
I1	17/05/2007	0	0	0	3394	424	0	0	0	0	520	0	0	0	0	0	0

Lake	Date	Microcystis sp.	Oscillatoria redekei	Spirulina	Cryptomonas spp.	Rhodomonas spp.	Euglena sp.	Euglena acus	Phacus sp.	Phacus caudatum	Trachelomonas sp.	Trachelomonas bacillifera	Ceratium hirundinella	Gymnodinium sp.	Peridinium sp.	Dinobryon cylindricum var. alpinum	Mallomonas sp.
I1	14/06/2007	1233	0	0	2023	890	0	0	0	0	0	0	0	0	0	0	0
I1	16/07/2007	0	0	0	1956	2140	0	0	0	0	0	0	0	0	0	0	0
I1	08/08/2007	0	0	0	19228	36247	0	0	0	0	0	0	0	0	0	0	0
I1	11/09/2007	0	0	0	36416	6992	0	0	0	0	0	0	0	0	0	#####	0
I1	15/10/2007	0	0	0	0	1981	3075	0	0	0	5614	0	0	0	0	0	0
I1	15/11/2007	0	0	0	120173	30892	0	0	0	0	1146112	0	0	0	0	0	0
I1	13/12/2007	0	0	0	23873	19705	0	0	0	0	2416	0	0	0	0	0	0
I1	17/01/2008	0	0	0	13757	3305	0	0	0	0	0	0	0	0	0	0	263
I1	19/02/2008	0	0	0	13656	3146	0	0	884	0	6988	0	0	0	0	0	2363
I1	18/03/2008	0	0	0	43160	2712	0	0	0	0	2761	0	0	0	0	0	8401
I2	22/03/2005	0	0	0	23718	3306	0	0	0	0	0	0	0	142	0	0	0
I2	15/04/2005	0	0	0	10242	2486	0	0	0	0	0	0	0	396	0	0	0
I2	11/05/2005	0	0	0	791	4790	0	0	0	0	0	0	0	0	0	0	0
I2	09/06/2005	0	0	0	9631	93801	0	0	0	0	0	0	0	0	0	0	3749
I2	06/07/2005	0	0	0	739	77	0	0	0	0	0	0	0	0	0	0	0
I2	01/08/2005	0	0	0	7688	8613	396	0	0	0	863	0	0	258	2201	0	0
I2	31/08/2005	9860	0	0	148093	22883	0	0	0	0	0	0	0	689	0	0	0
I2	04/10/2005	0	0	0	11270	1789	0	0	0	0	665	0	0	0	0	0	0
I2	01/11/2005	0	0	0	1052	1042	0	0	0	0	83	0	0	0	0	0	0
I2	24/11/2005	0	0	0	2529	9344	924	0	0	0	0	0	0	0	0	0	0
I2	21/12/2005	0	0	0	0	7691	3169	0	0	0	0	0	0	0	0	0	0
I2	17/01/2006	0	0	0	5564	5117	0	0	0	0	0	0	0	0	0	0	0
I2	15/02/2006	0	0	0	46532	11569	0	0	0	0	0	9663	0	0	0	0	0
I2	15/03/2006	0	0	0	37630	12713	0	0	0	0	1380	0	0	0	0	0	0
I2	20/04/2006	0	0	0	66035	16883	0	0	0	0	2209	0	0	0	0	0	0
I2	18/05/2006	0	0	0	66590	13948	0	0	0	0	0	0	0	0	0	0	0
I2	15/06/2006	0	0	0	185	1483	34	0	0	0	22	0	0	0	0	0	0
I2	13/07/2006	0	0	0	27316	8859	1150	0	0	0	0	0	4985	750	4793	0	0
I2	10/08/2006	47329	0	0	18775	610	282	0	0	0	2761	0	0	1102	0	0	0
I2	07/09/2006	2945	0	0	3162	272	9	0	0	0	155	0	0	0	0	0	0
I2	04/10/2006	0	0	0	3497	218	50	0	0	0	937	0	0	0	0	0	0
I2	02/11/2006	0	0	0	4689	4096	0	0	0	0	480	0	0	0	0	0	0
I2	30/11/2006	0	0	0	2072	16055	0	0	0	0	0	0	0	0	0	0	0
I2	20/12/2006	0	0	0	460	2277	0	0	0	0	65	0	0	0	0	0	0
I2	24/01/2007	0	0	0	2779	6291	0	0	0	0	0	0	0	0	521	0	0
I2	15/02/2007																
I2	15/03/2007	0	0	0	12498	2627	78	0	0	0	0	0	0	0	0	0	0
I2	19/04/2007	0	0	0	11150	2175	0	0	0	0	77	0	0	0	0	0	0
I2	17/05/2007	0	0	0	17264	2260	0	0	0	0	0	0	0	0	0	0	1867
I2	14/06/2007	0	0	0	839	2423	0	0	0	0	170	0	0	0	0	63	0
I2	16/07/2007	0	0	0	18343	3729	0	0	0	0	0	0	0	0	0	0	0
I2	08/08/2007	4382	0	0	61503	1695	0	0	0	0	614	0	0	0	0	0	0
I2	11/09/2007	0	0	0	18613	2684	0	0	0	0	383	0	0	0	0	0	0
I2	15/10/2007	0	0	0	2293	1780	0	0	0	0	0	0	0	0	0	0	0
I2	15/11/2007	0	0	0	9711	23773	0	0	0	0	0	0	0	0	0	0	0
I2	13/12/2007	0	0	0	553526	109838	0	0	0	0	0	0	0	0	0	0	0
I2	17/01/2008	0	0	0	59480	9662	0	0	0	0	4486	0	0	4133	0	0	5251
I2	19/02/2008	0	0	0	49785	4568	192	0	0	0	33820	0	0	0	0	0	0
I2	18/03/2008	0	0	0	2144186	410062	0	0	90548	0	238540	0	0	476107	0	0	0
I3	22/03/2005	0	0	0	34071	23624	511	0	3423	0	14029	0	0	1000	31953	0	25918
I3	15/04/2005	0	0	0	621503	86786	0	0	0	0	0	0	0	117557	0	0	268831
I3	11/05/2005	0	0	0	76042	1926	198	0	0	0	194	0	0	0	2469	0	0
I3	09/06/2005	0	0	0	243998	2893	0	0	0	0	0	0	0	0	0	0	0
I3	06/07/2005	0	0	0	29133	1144	264	0	0	0	1035	0	0	0	0	0	3150
I3	01/08/2005	21733	0	0	446	93	0	0	650	0	0	0	0	633	7277	0	0
I3	31/08/2005	18283	0	0	3376	3182	0	0	364	0	0	0	0	8621	0	0	0
I3	04/10/2005	18866	0	0	24774	973	0	0	0	0	0	0	0	0	0	0	8707
I3	01/11/2005	0	0	0	751197	5447	0	0	0	0	8214	0	0	0	0	0	0
I3	24/11/2005	4733	0	0	134400	488	0	0	0	0	0	0	0	0	0	0	0
I3	21/12/2005	0	0	0	3211	62	93	0	14	0	0	0	0	24	0	0	0
I3	17/01/2006	0	0	0	5395	1130	0	0	0	138	4233	0	0	0	0	0	0
I3	15/02/2006	0	0	0	213297	71637	0	0	0	319718	6273	3137	0	0	20004	0	0
I3	15/03/2006	0	0	0	132172	8859	0	0	0	0	18037	0	0	0	28758	0	0
I3	20/04/2006	0	0	0	33762	8221	0	0	0	0	38515	18718	0	718	0	0	0
I3	18/05/2006	0	0	0	2428	334	66	0	0	10	0	0	0	0	0	0	0
I3	15/06/2006	0	0	0	2967	170	29	0	0	0	173	58	0	0	0	0	0
I3	13/07/2006	0	0	0	370	3138	805	0	0	0	1262	0	0	0	0	0	0
I3	10/08/2006	29581	0	0	4046	254	704	0	786	0	690	0	0	0	0	0	0
I3	07/09/2006	78882	0	0	80925	3051	1409	0	0	1868	11044	0	0	0	193685	0	0
I3	04/10/2006	42945	0	0	51694	4799	511	0	1712	0	0	0	0	0	0	0	6098
I3	02/11/2006	2684	0	0	57275	1384	96	0	0	42	564	0	0	0	0	0	0
I3	30/11/2006	478	0	213	7493	592	17	0	0	0	67	0	0	0	0	0	0
I3	20/12/2006	0	0	0	1517	1025	0	0	0	0	86	0	0	0	0	0	0

Lake	Date	Microcystis sp.	Oscillatoria redekei	Spirulina	Cryptomonas spp.	Rhodomonas spp.	Euglena sp.	Euglena acus	Phacus sp.	Phacus caudatum	Trachelomonas sp.	Trachelomonas bacillifera	Ceratium hirundinella	Gymnodinium sp.	Peridinium sp.	Dinobryon cylindricum var. alpinum	Mallomonas sp.
I3	24/01/2007	0	0	0	1770	1397	0	0	0	0	74	0	0	0	235	0	0
I3	15/02/2007	0	0	0	149161	11716	0	0	0	0	0	0	0	0	0	0	12097
I3	15/03/2007	0	0	0	14539	4014	0	0	0	0	1127	0	0	0	7190	0	0
I3	19/04/2007	0	0	0	60424	59213	0	0	0	0	0	0	0	0	0	0	0
I3	17/05/2007	0	0	0	149464	6645	0	0	0	0	0	0	0	0	0	0	0
I3	14/06/2007	0	0	0	570768	448319	0	0	0	0	0	0	0	0	0	0	0
I3	16/07/2007	0	0	0	127052	11187	0	0	0	0	0	0	0	0	13206	0	5251
I3	08/08/2007	0	0	0	9323	1465	0	0	2264	0	0	0	0	0	0	0	0
I3	11/09/2007	0	0	0	904287	93729	0	0	0	0	15903	0	0	15870	0	0	48390
I3	15/10/2007	0	0	0	38064	1216	140	0	0	0	3852	0	0	549	1755	0	1674
I3	15/11/2007	0	0	0	98324	1780	0	0	0	0	0	0	0	0	0	0	0
I3	13/12/2007	0	0	0	9711	85	0	0	0	0	115	0	0	38	0	0	0
I3	17/01/2008	0	0	0	1239	350	0	0	0	0	0	0	0	43	0	0	33
I3	19/02/2008	0	0	0	0	10848	0	0	0	0	0	0	0	0	0	0	0
I3	18/03/2008	0	0	0	671224	262440	0	0	7244	0	165388	0	0	12696	81136	0	19356

D: Zooplankton species

	Lake					
	C1	C2	C3	I1	I2	I3
Cladocera						
Alona sp.	X				X	X
Bosmina coregoni (Baird) s. str.			X			X
Bosmina longirostris (O. F. Müller)	X	X	X	X	X	X
Ceriodaphnia megalops Sars					X	
Ceriodaphnia sp.	X	X	X	X	X	X
Chydorus ovalis (Kurz)					X	
Daphnia cucullata Sars. s. str.	X		X		X	X
Daphnia curvirostris (Eylmann)	X			X		
Daphnia hyalina (Leydig)	X	X	X	X		
Daphnia hyalina var. galeata Sars			X			X
Daphnia hyalina var. lacustris Sars				X	X	X
Daphnia longispina (O. F. Müller)			X			
Daphnia pulex (De Geer)					X	
Eurycercus lamellatus (O. F. Müller)	X	X	X		X	X
Scapheloberis mucronata (O. F. Müller)						
Sida crystallina (O. F. Müller)	X		X	X	X	
Copepods						
Cyclopoid copepods	X	X	X	X	X	X
Calanoid copepods	X	X	X	X	X	X
Rotifers						
Keratella spp.	X	X	X	X	X	X
Asplancha spp.	X	X	X	X	X	X
Ostracoda						
Un-indentified ostracod					X	

E: Zooplankton dataset

(All data presented as individuals L^{-1}).

Lake	Date	Total individuals	Ceriodaphnia sp.	Ceriodaphnia megalops	Daphnia cullata	Daphnia curvirostris	Daphnia hyalina var. gellata	Daphnia hyalina var. lacustris	Daphnia hyalina	Daphnia longispora	Daphnia pulex	Calanoid copepods	Cyclopoid copepods	Bosmina coregoni	Bosmina longiostris	Sida sp.	Sida crystallina	Chydorus ovalis	Eurycercus lamellatus	Alona spp.	Asplancha spp.	Keratella spp.	Ostracod
C1	22/03/2005	6.7	0.1	0.0	1.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.1	0.0	0.7	0.0	0.0	0.0	0.1	0.0	0.1	0.0	0.0
C1	15/04/2005	12.6	0.0	0.0	0.0	0.0	0.0	0.0	4.6	0.0	0.0	0.0	3.7	0.0	4.2	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
C1	12/05/2005	4.9	0.0	0.0	0.0	0.0	0.0	0.0	3.2	0.0	0.0	0.0	1.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C1	09/06/2005	9.1	0.0	0.0	0.0	0.0	0.0	0.0	6.3	0.0	0.0	0.0	2.8	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C1	06/07/2005	13.9	0.0	0.0	0.0	0.0	0.0	0.0	1.8	0.0	0.0	0.0	11.2	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
C1	01/08/2005	25.2	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	24.6	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
C1	31/08/2005	9.2	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	7.7	0.0	1.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C1	04/10/2005	32.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.0	0.0	18.5	0.0	12.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C1	01/11/2005	2.4	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.6	0.0	1.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C1	24/11/2005	1.7	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.2	0.0	1.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C1	21/12/2005	0.9	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.3	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C1	17/01/2006	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.2	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C1	16/02/2006	2.7	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	2.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
C1	15/03/2006	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.1	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C1	20/04/2006	2.2	0.0	0.0	0.0	0.0	0.0	0.0	0.9	0.0	0.0	0.0	0.5	0.0	0.7	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0
C1	18/05/2006	2.4	0.0	0.0	0.0	0.0	0.0	0.0	1.8	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C1	15/06/2006	13.2	0.0	0.0	0.0	0.0	0.0	0.0	9.2	0.0	0.0	0.1	2.8	0.0	1.1	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
C1	13/07/2006	55.5	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	35.9	0.0	19.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C1	10/08/2006	23.8	0.1	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	16.1	0.0	7.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
C1	07/09/2006	189.1	0.0	0.0	0.0	0.0	0.0	0.0	2.4	0.0	0.0	0.0	16.2	0.0	170.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C1	05/10/2006	22.2	0.1	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	1.1	0.0	18.4	0.0	0.0	0.0	0.0	0.0	1.9	0.0	0.0
C1	02/11/2006	3.4	0.0	0.0	0.0	0.0	0.0	0.0	1.2	0.0	0.0	0.0	0.9	0.0	0.1	0.0	0.0	0.0	0.0	0.0	1.2	0.0	0.0
C1	30/11/2006	5.4	0.0	0.0	0.0	0.0	0.0	0.0	4.5	0.0	0.0	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
C1	20/12/2006	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C1	24/01/2007	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C1	15/02/2007	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C1	15/03/2007	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C1	19/04/2007	15.9	0.0	0.0	0.0	0.0	0.0	0.0	9.3	0.0	0.0	0.0	4.1	0.0	1.4	1.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0
C1	17/05/2007	6.6	0.0	0.0	0.0	0.0	0.0	0.0	5.7	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.4	0.0	0.0
C1	14/06/2007	22.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	8.6	0.0	0.1	0.1	0.0	0.0	0.0	0.0	12.7	0.0	0.0
C1	16/07/2007	2.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	1.4	0.0	0.0	0.0	0.0	0.0	0.8	0.1	0.0
C1	08/08/2007	23.5	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	1.8	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	20.9	0.0
C1	11/09/2007	6.8	0.4	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	1.0	0.0	2.4	0.0	0.0	0.0	0.0	0.0	2.6	0.0	0.0
C1	15/10/2007	17.2	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	2.4	0.0	14.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0
C1	15/11/2007	2.8	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.5	0.0	1.1	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
C1	13/12/2007	1.6	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.2	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C1	17/01/2008	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C1	19/02/2008	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C1	18/03/2008	1.2	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0
C2	22/03/2005	16.2	0.0	0.0	0.0	0.0	0.0	0.0	3.2	0.0	0.0	0.0	8.6	0.0	2.3	0.0	0.0	0.0	0.3	0.0	1.8	0.0	0.0
C2	15/04/2005	5.4	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	1.2	0.0	1.9	0.0	0.0	0.0	0.0	0.0	1.3	0.0	0.0
C2	12/05/2005	6.1	0.0	0.0	0.0	0.0	0.0	0.0	3.1	0.0	0.0	0.0	2.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C2	09/06/2005	5.0	0.0	0.0	0.0	0.0	0.0	0.0	2.9	0.0	0.0	0.0	2.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C2	06/07/2005	25.5	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	25.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
C2	01/08/2005	57.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	56.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.0
C2	31/08/2005	20.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	18.7	0.0	1.7	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0
C2	04/10/2005	18.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	13.3	0.0	4.7	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0
C2	01/11/2005	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
C2	24/11/2005	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
C2	21/12/2005	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.2	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C2	17/01/2006	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C2	16/02/2006	4.3	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	3.4	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C2	15/03/2006	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C2	20/04/2006	5.9	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0	1.3	0.0	1.4	0.0	0.0	0.0	0.0	0.0	0.8	0.3	0.0
C2	18/05/2006	3.9	0.0	0.0	0.0	0.0	0.0	0.0	2.3	0.0	0.0	0.0	1.2	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
C2	15/06/2006	9.4	0.0	0.0	0.0	0.0	0.0	0.0	5.9	0.0	0.0	0.0	1.0	0.0	2.4	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
C2	13/07/2006	42.4	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	39.9	0.0	1.9	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0
C2	10/08/2006	45.4	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	39.1	0.0	5.7	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0
C2	07/09/2006																						

Lake	Date	Total individuals	Ceriodaphnia sp.	Ceriodaphnia megalops	Daphnia cullata	Daphnia curvirostris	Daphnia hyalina var. gellata	Daphnia hyalina var. lacustris	Daphnia hyalina	Daphnia longispora	Daphnia pulex	Calanoid copepods	Cyclopoid copepods	Bosmina coregoni	Bosmina longiostris	Sida sp.	Sida crystallina	Chydorus ovalis	Eurycercus lamellatus	Alona spp.	Asplancha spp.	Keratella spp.	Ostracod
C2	15/10/2006	17.9	0.2	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0	0.0	3.4	0.0	12.1	0.0	0.0	0.0	0.0	0.0	1.5	0.0	0.0
C2	02/11/2006	1.5	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.3	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
C2	30/11/2006	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C2	20/12/2006	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C2	24/01/2007	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C2	15/02/2007	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C2	15/03/2007	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
C2	19/04/2007	13.3	0.0	0.0	0.0	0.0	0.0	0.0	5.9	0.0	0.0	0.0	3.7	0.0	3.2	0.0	0.0	0.0	0.0	0.0	0.5	0.1	0.0
C2	17/05/2007	1.9	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.1	0.0	0.1	0.0	0.0	0.0	0.1	0.0	1.1	0.0	0.0
C2	14/06/2007	1.2	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
C2	16/07/2007	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C2	08/08/2007	7.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	1.8	0.0	0.0	0.0	0.0	0.0	4.8	0.6	0.0
C2	11/09/2007	5.5	0.1	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	3.2	0.0	0.4	0.0	0.0	0.0	0.0	0.0	1.8	0.0	0.0
C2	15/10/2007	7.5	0.1	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	3.0	0.0	2.9	0.0	0.0	0.0	0.0	0.0	1.5	0.1	0.0
C2	15/11/2007	2.9	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.8	0.0	2.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
C2	13/12/2007	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C2	17/01/2008	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C2	19/02/2008	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C2	18/03/2008	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C3	22/03/2005	52.3	0.0	0.0	0.3	0.0	0.0	0.0	1.8	0.0	0.0	0.0	39.6	10.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C3	15/04/2005	6.6	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	4.6	0.0	1.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C3	12/05/2005	4.6	0.0	0.0	0.0	0.0	0.0	0.0	1.1	0.0	0.0	0.0	3.4	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C3	09/06/2005	1.4	0.0	0.0	0.0	0.0	0.0	0.0	0.9	0.1	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C3	06/07/2005	27.1	0.0	0.0	0.0	0.0	0.0	0.0	3.6	0.0	0.0	0.0	23.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C3	01/08/2005	26.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	26.0	0.0	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C3	31/08/2005	39.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	37.2	0.0	2.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C3	04/10/2005	16.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.1	0.0	0.0	13.4	0.0	1.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C3	01/11/2005	3.4	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.4	0.0	2.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C3	24/11/2005	2.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	2.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C3	21/12/2005	1.3	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.2	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C3	17/01/2006	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C3	16/02/2006	2.2	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	1.4	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C3	15/03/2006	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.4	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C3	20/04/2006	9.8	0.0	0.0	0.0	0.0	0.0	0.0	1.2	0.0	0.0	0.0	0.2	0.0	2.3	0.0	0.0	0.0	0.7	0.0	5.3	0.1	0.0
C3	18/05/2006	2.1	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.5	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0
C3	15/06/2006	8.0	0.0	0.0	0.0	0.0	0.0	0.0	5.5	0.0	0.0	0.0	0.8	0.0	0.6	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0
C3	15/07/2006																						
C3	10/08/2006	44.8	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	39.1	0.0	5.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C3	07/09/2006	90.1	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0	66.0	0.0	20.7	0.0	0.0	0.0	0.0	0.0	1.4	0.0	0.0
C3	05/10/2006	21.1	0.2	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	7.8	0.0	12.0	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0
C3	02/11/2006	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C3	30/11/2006	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C3	20/12/2006	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C3	24/01/2007	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C3	15/02/2007	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C3	15/03/2007	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
C3	19/04/2007	12.9	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0	9.3	0.0	0.3	0.3	0.0	0.0	0.0	0.0	0.9	0.0	0.0
C3	17/05/2007	4.5	0.0	0.0	0.0	0.0	0.0	0.0	0.9	0.0	0.0	0.0	0.4	0.0	0.1	0.0	0.0	0.0	0.0	0.0	3.0	0.0	0.0
C3	16/06/2007	36.2	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	25.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	10.5	0.0	0.0
C3	16/07/2007	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C3	08/08/2007	4.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	1.1	0.0	0.0	0.0	0.0	0.0	3.1	0.3	0.0
C3	11/09/2007	18.2	0.5	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	9.5	0.0	6.6	0.0	0.0	0.0	0.0	0.0	1.5	0.0	0.0
C3	15/10/2007	19.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.2	0.0	17.1	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0
C3	15/11/2007	3.2	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.4	0.0	2.4	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0
C3	13/12/2007	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C3	17/01/2008	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C3	19/02/2008	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C3	18/03/2008	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I1	22/03/2005	27.0	0.0	0.0	0.0	0.0	0.0	0.0	21.4	0.0	0.0	0.5	5.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Lake	Date	Total individuals	Ceriodaphnia sp.	Ceriodaphnia megalops	Daphnia cullata	Daphnia curvirostris	Daphnia hyalina var. gellata	Daphnia hyalina var. lacustris	Daphnia hyalina	Daphnia longispora	Daphnia pulex	Calanoid copepods	Cyclopoid copepods	Bosmina coregoni	Bosmina longiostris	Sida sp.	Sida crystallina	Chydorus ovalis	Eurycercus lamellatus	Alona spp.	Asplancha spp.	Keratella spp.	Ostracod
I1	15/04/2005	1.3	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0	0.0	0.2	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I1	12/05/2005	5.2	0.0	0.0	0.0	0.0	0.0	0.0	4.5	0.0	0.0	0.0	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I1	09/06/2005	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.2	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I1	06/07/2005	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I1	01/08/2005	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I1	31/08/2005	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I1	04/10/2005	4.4	0.0	0.0	0.0	0.0	0.0	0.0	1.6	0.0	0.0	2.5	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I1	01/11/2005	6.4	0.0	0.0	0.0	0.0	0.0	0.0	4.8	0.0	0.0	1.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I1	24/11/2005	3.5	0.0	0.0	0.0	0.0	0.0	0.0	2.2	0.0	0.0	1.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I1	21/12/2005	2.5	0.0	0.0	0.0	0.0	0.0	0.0	1.9	0.0	0.0	0.5	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I1	17/01/2006	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I1	16/02/2006	5.2	0.0	0.0	0.0	0.0	0.0	0.0	4.5	0.0	0.0	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I1	15/03/2006	2.4	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	1.2	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I1	20/04/2006	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I1	18/05/2006	6.0	0.0	0.0	0.0	0.0	0.0	0.0	3.8	0.0	0.0	0.1	1.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0
I1	15/06/2006																						
I1	13/07/2006	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I1	10/08/2006	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I1	07/09/2006	3.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	2.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I1	05/10/2006	77.6	0.0	0.0	0.0	0.0	0.0	0.0	3.0	0.0	0.0	0.0	45.2	0.0	24.6	0.0	0.0	0.0	0.0	0.0	4.7	0.0	0.0
I1	02/11/2006	11.1	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	4.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.7	0.0	0.0
I1	30/11/2006	35.4	0.0	0.0	0.0	0.0	0.0	0.0	9.9	0.0	0.0	0.0	8.9	0.0	16.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I1	20/12/2006	5.0	0.0	0.0	0.0	0.0	0.0	0.0	3.6	0.0	0.0	0.0	0.8	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I1	24/01/2007	0.5	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.2	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
I1	15/02/2007	5.2	0.0	0.0	0.0	0.0	0.0	0.0	2.8	0.0	0.0	0.0	2.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
I1	15/03/2007	4.9	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0	1.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0
I1	19/04/2007	6.0	0.0	0.0	0.0	0.0	0.0	0.0	3.3	0.0	0.0	0.0	1.8	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.7	0.0	0.0
I1	17/05/2007	8.5	0.0	0.0	0.0	0.0	0.0	0.0	1.7	0.0	0.0	0.0	4.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0
I1	14/06/2007	2.4	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.2	1.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
I1	16/07/2007	3.8	0.1	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0	3.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I1	08/08/2007	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.2	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I1	11/09/2007	2.5	0.3	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.1	0.5	0.0	1.3	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0
I1	15/10/2007	4.8	0.0	0.0	0.0	0.0	0.0	0.0	3.3	0.0	0.0	0.1	1.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I1	15/11/2007	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.1	0.6	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I1	13/12/2007	8.8	0.0	0.0	0.0	0.0	0.0	0.2	0.6	0.0	0.0	0.1	7.7	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I1	17/01/2008	6.3	0.0	0.0	0.0	0.7	0.0	0.5	1.8	0.0	0.0	0.3	2.8	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I1	19/02/2008	1.2	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0	0.1	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
I1	18/03/2008	7.7	0.0	0.0	0.0	0.0	0.0	4.5	2.4	0.0	0.0	0.2	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0
I2	22/03/2005																						
I2	15/04/2005	6.2	0.0	0.0	0.0	0.0	0.0	0.0	1.7	0.0	0.0	1.9	2.4	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I2	12/05/2005	7.2	0.0	0.0	0.0	0.0	0.0	0.0	2.6	0.0	0.1	2.3	2.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I2	09/06/2005	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I2	06/07/2005	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I2	01/08/2005	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I2	31/08/2005	8.4	2.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.7	0.0	2.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I2	04/10/2005	18.4	5.7	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.2	5.3	0.0	6.7	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
I2	01/11/2005	7.6	0.0	0.0	0.0	0.0	0.0	0.0	6.6	0.0	0.0	0.3	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I2	24/11/2005	5.1	0.0	0.0	0.0	0.0	0.0	0.3	4.1	0.0	0.0	0.4	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I2	21/12/2005	4.3	0.0	0.0	0.0	0.0	0.0	0.0	3.2	0.0	0.0	0.8	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I2	17/01/2006	2.0	0.0	0.0	0.0	0.0	0.0	0.0	1.3	0.0	0.0	0.5	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I2	16/02/2006	3.8	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	1.3	1.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
I2	15/03/2006	2.9	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0	0.2	1.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
I2	19/04/2006	6.0	0.0	0.0	0.0	0.0	0.0	0.0	2.5	0.0	0.0	0.1	1.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.6	0.0	0.0
I2	18/05/2006	2.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.4	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
I2	15/06/2006	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I2	13/07/2006	1.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.5	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I2	10/08/2006	7.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	1.4	4.1	0.0	0.3	0.2	0.0	0.0	0.0	0.0	0.4	0.2	0.0
I2	07/09/2006	290.6	0.0	0.0	196.9	0.0	0.0	0.0	0.0	0.0	0.0	17.8	61.9	0.0	13.1	0.0	0.0	0.0	0.0	0.0	0.9	0.0	0.0
I2	05/10/2006	5.0	0.5	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	2.4	0.8	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0

Lake	Date	Total individuals	Ceriodaphnia sp.	Ceriodaphnia megalops	Daphnia cullata	Daphnia curvirostris	Daphnia hyalina var. gellata	Daphnia hyalina var. lacustris	Daphnia hyalina	Daphnia longispora	Daphnia pulex	Calanoid copepods	Cyclopoid copepods	Bosmina coregoni	Bosmina longiostris	Sida sp.	Sida crystallina	Chydorus ovalis	Eurycercus lamellatus	Alona spp.	Asplancha spp.	Keratella spp.	Ostracod
I2	02/11/2006	6.4	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	5.5	0.2	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0
I2	30/11/2006	6.7	0.1	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0	5.8	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I2	20/12/2006	3.2	0.0	0.0	0.0	0.0	0.0	0.0	0.9	0.0	0.0	2.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I2	24/01/2007	0.9	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I2	15/02/2007	0.7	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I2	15/03/2007	7.7	0.0	0.0	0.0	0.0	0.0	0.4	6.0	0.0	0.0	0.5	0.2	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.3	0.3	0.0
I2	19/04/2007	13.2	0.0	0.0	0.0	0.0	0.0	0.2	8.9	0.0	0.0	1.0	2.1	0.0	0.1	0.1	0.0	0.0	0.0	0.0	0.6	0.0	0.0
I2	17/05/2007	3.4	0.0	0.0	0.0	0.0	0.0	0.1	1.3	0.0	0.0	0.2	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.2	0.0
I2	14/06/2007	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.5	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0
I2	16/07/2007	30.6	2.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	22.9	4.1	0.0	0.6	0.0	0.0	0.1	0.2	0.0	0.0	0.0	0.1
I2	08/08/2007	35.1	28.8	2.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	3.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
I2	11/09/2007	23.4	18.2	0.1	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	2.2	0.0	2.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I2	15/10/2007	2.1	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	1.2	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I2	15/11/2007	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I2	13/12/2007	2.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I2	17/01/2008	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I2	19/02/2008	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I2	18/03/2008	1.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	1.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I3	22/03/2005																						
I3	15/04/2005	20.7	0.0	0.0	0.0	0.0	0.0	0.0	3.6	0.0	0.0	0.0	16.9	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
I3	12/05/2005	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I3	09/06/2005	0.9	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.0
I3	06/07/2005	12.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	12.2	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
I3	01/08/2005	1.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.3	0.5	0.0
I3	31/08/2005	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I3	04/10/2005	17.4	0.7	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	15.9	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0
I3	01/11/2005	6.3	0.0	0.0	1.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I3	24/11/2005	10.5	0.1	0.0	6.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0
I3	21/12/2005	13.3	0.0	0.0	7.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.3	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0
I3	17/01/2006	2.9	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.8	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
I3	16/02/2006	8.5	0.0	0.0	0.0	0.0	0.0	0.0	1.6	0.0	0.0	0.2	6.2	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0
I3	15/03/2006	1.4	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
I3	20/04/2006	6.3	0.0	0.0	0.0	0.0	0.0	0.0	1.9	0.0	0.0	0.3	3.3	0.5	0.0	0.0	0.0	0.0	0.1	0.0	0.2	0.0	0.0
I3	18/05/2006	40.2	0.0	0.0	0.0	0.0	0.0	0.0	11.4	0.0	0.0	1.5	23.3	0.0	2.5	0.0	0.0	0.0	1.5	0.0	0.0	0.0	0.0
I3	15/06/2006	29.0	0.0	0.0	22.6	0.0	0.0	0.0	1.8	0.0	0.0	0.2	2.7	0.0	1.2	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0
I3	13/07/2006	4.4	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.1	2.0	0.0	1.4	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0
I3	10/08/2006	2.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.9	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.3	0.2	0.0
I3	07/09/2006	5.4	0.6	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.8	1.4	0.0	0.1	0.0	0.0	0.0	0.0	0.0	2.3	0.0	0.0
I3	05/10/2006	3.5	2.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.1	1.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0
I3	02/11/2006	9.0	0.1	0.0	6.2	0.0	0.0	0.0	0.0	0.0	0.0	0.6	1.5	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0
I3	30/11/2006	6.9	0.0	0.0	3.7	0.0	0.0	0.0	0.0	0.0	0.0	0.8	1.3	0.0	0.9	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0
I3	20/12/2006																						
I3	24/01/2007	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I3	15/02/2007	1.3	0.2	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.2	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I3	15/03/2007	1.4	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.2	0.6	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I3	19/04/2007	48.5	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.5	16.4	0.0	23.8	0.0	0.0	0.0	0.0	0.0	6.9	0.0	0.0
I3	17/05/2007	4.6	0.0	0.0	0.0	0.0	0.0	0.0	2.6	0.0	0.0	0.8	1.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I3	14/06/2007	17.3	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.0	0.0	15.1	0.0	1.2	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
I3	16/07/2007	5.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.1	0.0	0.5	0.0	0.0	0.0	0.0	0.0	1.4	0.0	0.0
I3	08/08/2007	7.0	0.3	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	1.7	0.0	0.3	0.0	0.0	0.0	0.0	0.0	4.2	0.0	0.0
I3	11/09/2007	38.8	0.4	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.0	0.0	23.5	0.0	0.4	0.0	0.0	0.0	0.0	0.0	13.8	0.0	0.0
I3	15/10/2007	39.8	0.0	0.0	0.0	0.0	0.4	0.0	1.5	0.0	0.0	0.0	17.1	0.0	20.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I3	15/11/2007	13.5	0.0	0.0	8.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0	2.6	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0
I3	13/12/2007	6.5	0.0	0.0	0.0	0.0	0.0	1.5	3.2	0.0	0.0	0.0	1.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
I3	17/01/2008	0.9	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
I3	19/02/2008	0.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
I3	18/03/2008	2.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.8	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0